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SUMMARY OF AVAILABLE INFORMATION ON CHESAPEAKE BAY SUBMERGED VEGETATION

MARYLAND DEPARTMENT OF NATURAL RESOURCES,
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Submerged aquatic species tend to inhabit the shallow, shoreline areas of the Bay and its subestuaries, primarily limited to depths of three meters or less. Species vary as to salinity and temperature tolerances, morphology, preferred bottom substrate, susceptibility to chemical pollutants and general distribution. There are approximately eleven species of submerged aquatic vegetation (SAV) dominant in the waters of the Chesapeake Bay.

Submerged aquatic flora constitutes the principal source of food for waterfowl and some fish; they provide direct or indirect food and shelter for many of the small host organisms that are eaten by fish and other predators. The spawning activities of certain organisms require SAV: they purify the water by removing various noxious substances and returning oxygen; they shade the underlying waters and sediments from solar heating; and they provide an important source of detritus. Submerged aquatics help stabilize sediments and reduce shoreline erosion.

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i (b)

PREFACE

The technical document Summary of Available Information on Chesapeake Bay Submerged Vegetation has been prepared by the University of Maryland Horn Point Environmental Laboratory in order to provide an initial data base on submerged aquatic vegetation in the Chesapeake Bay. Available information has been segregated into six main categories: biological summaries of dominant plant species; environmental values of aquatic plants; historic documentation; factors related to aquatic plant occurrence and growth; modelling; and management options. This technical document is a literature search and is designed to serve as a reference document for the Chesapeake Bay.

Any suggestions or questions regarding this summary should be directed to:

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EXECUTIVE SUMMARY

There are approximately eleven species of submerged aquatic vegetation (SAV) dominant in the waters of the Chesapeake Bay. Included in this group and discussed in this technical document are:

Potamogeton _perfoliatus_ - redhead grass
Ruppia _maritima_ - widgeon grass
Myriophyllum _spicatum_ - Eurasian watermilfoil
Zostera _marina_ - eelgrass
Potamogeton _pectinatus_ - sago pondweed
Zannichellia _palustris_ - horned pondweed
Vallisneria _americana_ - wildcelery
Elodea _canadensis_ - common elodea
Chara spp. - muskgrass
Ceratophyllum _demersum_ - hornwort or coontail
Najas spp. - naiads

Submerged aquatic species tend to inhabit the shallow, shoreline areas of the Bay and its subestuaries, primarily limited to depths of three meters or less. Species vary as to salinity and temperature tolerances, morphology, preferred bottom substrate, susceptibility to chemical pollutants and general distribution.

Submerged aquatic flora constitutes the principal source of food for waterfowl and some fish; they provide direct or indirect food and shelter for many of the small host organisms that are eaten by fish and other predators. The spawning activities of certain organisms require SAV: they purify the water by removing various noxious substances and returning oxygen; they shade the underlying waters and sediments from solar heating; and they provide an important source of detritus. The ability to assimilate inorganic substances into organic compounds usable by other organisms enhances the importance of these plants as a vital link in the food chain. Functioning as prime areas for hiding and breeding, this vegetation group provides surfaces for the attachment of eggs. Also, an abundance of animal life, including insect, crustacean, molluscan, fish and others find excellent feeding grounds on and around SAV.

Submerged aquatics help stabilize sediments and reduce shoreline erosion. Bays with healthy benthic flora have been shown to have relatively stable metabolism with less fluctuation in comparison to plankton dominated bays. SAV further functions in the estuarine nutrient cycle as a trap for dissolved phosphorus and nitrogen. Though not commonly viewed as indicators of biological conditions or trends, SAV is considered to be potentially useful as an indicator of such pollution trends.

Documentation of historic distribution and abundance of SAV within the Bay estuary begins in the late 1800s, though information remains sparse until the late 1950s when the first comprehensive survey was initiated on the Susquehanna Flats. Since that time other surveys have been conducted over both small areas of the Bay and the entire Maryland portion. Based on the results of these surveys plus herbaria records, newspaper accounts and personal observations, the conclusion was reached that submerged aquatic vegetation in the Bay has not only declined but changed in species density, diversity and distribution. Some native species have disappeared, dominance patterns have changed, drastic fluctuations in yearly populations have been noted and seasonal patterns of growth, decline and reproduction have shifted.

In order to determine the probable cause or causes for the changing patterns in submerged vegetation, the various factors that are known to affect the grasses have been analyzed to the extent possible given the availability of published and unpublished literature. Included among these factors are: agrochemicals, turbidity, salinity, temperature, pH, wave action, fauna, epiphytes, bicarbonate ion, chlorine, disease, boat traffic, dredging, nutrient loading, petroleum and heavy metals. In order to assess these impacts and correlate them to a Baywide decline in submerged grasses, environmental factors can be initially separated into short-term, localized impacts or factors that impact the Bay as a whole or on an aggregate basis such as the upper, middle or lower Bay areas.

Factors which are known to be localized or short-term are : turbidity, dredging, nutrient loading and possibly chlorine. Data relating to turbidity are available only for the past ten years but based on this time span, there does not appear to be a Baywide increase in turbidity. Some subestuaries such as the Sassafras River and Eastern Bay have documented increases in suspended sediment, but others such as the Magothy, Chester and Patuxent Rivers show turbidity decreases. Average annual Secchi disc readings for the Maryland Bay indicate a 1.25 cm increase in light penetration from 1972 to 1976. Increasing human activity in the Bay area would be expected to result in more turbid water conditions with less light penetration. Based on this apparent discrepancy, turbidity has been viewed as a localized impact and not a Baywide phenomenon. Thus there is no support for attributing Baywide grass losses solely to increasing turbidity.

There is some experimental evidence that chlorine and/or its by-products negatively affect SAV; however, the fate and mode of action of chlorine in an aquatic environment is largely unknown. Chlorine has been implicated along with excessive nutrient loading in the upper Potomac River to cause the extensive losses of SAV over the past 50 years. Municipal and industrial use of chlorine has escalated dramatically since 1900, resulting in increased usage of Bay waters as a repository for chlorinated and nutrient rich effluent. There are no data available on chlorine residual levels in the Bay that are sufficiently comprehensive to correlate with grass losses. If chlorine is to be considered as a negative factor in relation to SAV declines in the Bay, it is probable that chlorine is at least of local importance in areas receiving excessive municipal and industrial effluents.

Dredging and boat traffic result in benthic destruction and turbidity and are confined to localized areas of excessive water oriented activity. Due to their localized nature, either of these impacts could be responsible for SAV losses only in specific areas.

Factors that are applicable to the entire Bay or a large portion of the Bay include: salinity, temperature, pH, wave action, fauna, epiphytes, bicarbonate ion, disease, agrochemicals, heavy metals and petroleum products. Salinity fluctuations have been regarded as a possible cause for Bay grass declines due to increasing salinities in the 1960s and decreasing salinities in the 1970s. Rising salinities may have negatively affected SAV; however, most species have a rather wide tolerance range that would allow for fairly substantial salinity increases without reaching toxic levels. The salinity decreases that are apparent during the 1970s would tend to enhance SAV growth and and reproduction for most species except Zostera marina which is found most frequently in near ocean salinities.

Some researchers have regarded rising temperatures as being responsible for SAV declines. Research indicates that while temperature increases do negatively affect submerged grasses and their reproduction, no Baywide increase in ambient temperature levels has been noted, at least in the Maryland portion of the Bay. Drastic fluctuations in pH can negatively impact submerged vegetation but such changes have not been evident in the Bay. The pH range of the Bay is relatively stable and given an otherwise health ecosystem, SAV is capable of adapting to pH fluctuations.

Wave action such as might be experienced by excessive storm events can impact submerged grasses. Such water movement is part of the dynamic nature of a wind driven estuary such as the Bay and thus is considered part of a natural cycle of events. The Bay has proved to be resilient to such impacts in the past so that given a normal recovery period, there is no reason to assume that the Bay will not continue to recycle subsequent to such events.

Faunal and epiphytic organisms are part of a normal ecosystem. There have been instances where the cownose ray (Rhinoptera bonasus), carp (Cyprinus carpio), and mute swan (Cygnus olor) have been known to uproot or overgraze SAV beds in specific areas, but there is no evidence of a large scale impact. This is also true for diseases. Zostera marina and Myriophyllum spicatum have declined in the past possibly due to a pathogenic organism or disease situation. Though these impacts have been noted on a large scale, they have not affected all species of submerged aquatics. In fact, the decline of M. spicatum in the 1960s resulted in increases in other SAV species.

The importance of the bicarbonate ion as a carbon source, especially in areas of low salinity, has not been conclusively established. Research is presently underway, but until evidence is brought forth, decreases in HCO_3^- can only be assumed to negatively impact SAV. The availability of HCO_3^- is directly related to salinity--the higher the salinity the more available the bicarbonate ion. Thus, if this factor has affected SAV losses, it would be relegated to the upper and less saline reaches of the Chesapeake Bay.

Changing farm techniques and cropping practices have resulted in substantial increases in the use of herbicides, fertilizers and lime. Herbicides appear to be detrimental to submerged aquatic vegetation. Recent bioassay work on Zannichellia palustris has shown that this species is highly susceptible to various levels of atrazine, linuron and trifluralin in the dissolved and suspended sediment portion of the water column plus in bottom sediments. Based on this evidence, there are indications that inhibitory herbicide levels may be involved in Bay grass declines. However, field oriented data are necessary in order to conclude that herbicides are a primary factor in SAV losses. The highly toxic and lasting nature of herbicides adsorbed to sediment particles appears to be a very important factor that requires further research.

Some heavy metals are known to accumulate in various species of submerged aquatics. There is some indication that these metals may have a negative impact on SAV but experimental data are not available to determine the extent of the impact nor are heavy metal levels in the Bay known. The same lack of experimental evidence exists regarding petroleum product impacts to SAV.

Once the relative impacts of the known factors applicable to SAV growth and development have been analyzed, it becomes possible to apply mathematical modeling techniques to the Bay estuarine system. After models are constructed and prove to be realistic, the management alternatives will be more clearly defined.

This technical document represents an effort to collect, synthesize and organize available information relating to submerged aquatic species dominant in the Chesapeake Bay and what environmental factors may be causing the recently noted SAV decline.

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LIST OF ABBREVIATIONS AND SYMBOLS

ABBREVIATIONS

Å	angstrom
a.e.	acid equivalent
a.i.	active ingredients
C	celsius
CBCES	Chesapeake Bay Center for Environmental Studies
CBL	Chesapeake Biological Laboratory
cm	centimeter
d	day
dm	decimeter
g	gram
g cal	gram calorie
ha	hectare
HPEL	Horn Point Environmental Laboratory
hr	hour
kg	kilogram
km	kilometer
kph	kilometer per hour
l	liter
m	meter
m^2	square meter
m^3	cubic meter
mm	millimeter
ml	milliliter
MHW	mean high water
MLW	mean low water
MBHRL	Migratory Bird and Habitat Research Laboratory
MW	megawatt
mμ	millimicron
nm	nanometer
NOAA	National Oceanic and Atmospheric Administration
PCB	polychlorinated biphenyl
ppb	parts per billion
ppm	parts per million
ppt	parts per thousand
r, r^2	correlation coefficient
SAV	submerged aquatic vegetation
sec	second
SES	steam electric station
SS	suspended solids
STP	sewage treatment plant
USDA	U.S. Department of Agriculture
USDI	U.S. Department of Interior
USSR	Union of Soviet Socialists Republic
UV	ultra violet
VIMS	Virginia Institute of Marine Science
μg	microgram
μE	micro Einstein

LIST OF ABBREVIATIONS AND SYMBOLS (cont.)

SYMBOLS

Br, Br^-	bromine, bromine ion
C, C^{++}	carbon, carbon ion
^{14}C	carbon 14
Ca, Ca^{++}	calcium, calcium ion
$CaCl_2$	calcium chloride
$CaCO_3$	calcium carbonate
$Ca(HCO_2)_2$	calcium formate
$Ca(OH)_2$	calcium hydroxide
CO_2	carbon dioxide
$^{14}CO_2$	labeled carbon dioxide
Cu	copper
$CuSO_4$	copper sulfate
H	hydrogen
HCl	hydrochloric acid
H_2CO_3	bicarbonate
HCO_3^-	bicarbonate ion
H_2S	hydrogen sulfide
Hg	mercury
$HgCl_2$	mercurous chloride
K	potassium
K_2CO_3	potassium carbonate
$KHCO_3$	potassium carbonate, acid
Mg, Mg^{++}	magnesium, magnesium ion
N	nitrogen
NH_4, NH_4^-	ammonia, ammonium ion
NH_4-N	ammonia nitrogen
Na, Na^+	sodium, sodium ion
NaCl	sodium chloride
$NaAsO_2$	sodium arsenite
$NaHCO_3$	sodium bicarbonate
N-P-K	nitrogen-phosphorus-potassium
NO_3, NO_3^-	nitrate, nitrate ion
NO_3-N	nitrate nitrogen
O_2	oxygen
P	phosphorus
TKN	total kjeldahl nitrogen

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CHAPTER 1

BIOLOGY

INTRODUCTION

The Chesapeake Bay has historically supported a wide variety of submerged aquatic vegetation (SAV). Over the years, there have been indications as to changes in species diversity, but the present populations consist of about ten dominant vascular hydrophytes and one species of macrophytic alga (Chara sp.). Chara is included because of its physical resemblance to the other species and its similar ecological values. Ulva and Enteromorpha are also considered important algal species native to the Chesapeake Bay but are not discussed in this technical document. Chara was initially chosen based on the results of the U.S. Fish and Wildlife Service Migratory Bird and Habitat Research Laboratory survey and the importance of Chara to waterfowl.

These eleven species of SAV inhabit the shallow, shoreline areas of the Bay primarily limited to depths of three meters or less. The most convenient classification system for these species is according to salinity tolerance. Najas, Chara and Vallisneria americana are fresh to slightly brackish water species found in the upper reaches of the Chesapeake Bay and in the fresh areas of the many subestuaries that comprise the vast Bay estuary system. Elodea canadensis, Myriophyllum spicatum and Ceratophyllum demersum tend to be found in more brackish areas while Potamogeton pectinatus, P. perfoliatus and Zannichellia palustris are tolerant of salinities up to about 20 ppt. Zostera marina and Ruppia maritima are capable of tolerating full ocean salinities, though Ruppia can inhabit not only marine conditions but also areas with a considerably lower salinity level. Consideration of salinity as a limiting factor is discussed in Chapter 2.

Information as to environmental requirements of individual species and their tolerances is not uniform from one species to another. Some species have been frequently utilized experimentally and have been extensively studied under field conditions. Others have received only cursory interest. This chapter attempts to present the most salient information available on each dominant Bay species in order to provide background data essential to the investigation of SAV declines in recent years.

POTAMOGETON PERFOLIATUS

Biology

General Vegetative Morphology. Potamogeton perfoliatus exhibits extensive morphological variation and has been separated into two varieties. Formerly, these two varieties were denoted as separate species, but have since been denoted as var. bupleuroides and var. richardsonii. Potamogeton perfoliatus var. bupleuroides will be reviewed in this section since it is the most common variety found in the Chesapeake Bay (Ogden 1943). For simplicity, this variety will be referred to as P. perfoliatus and is commonly known as redhead grass.

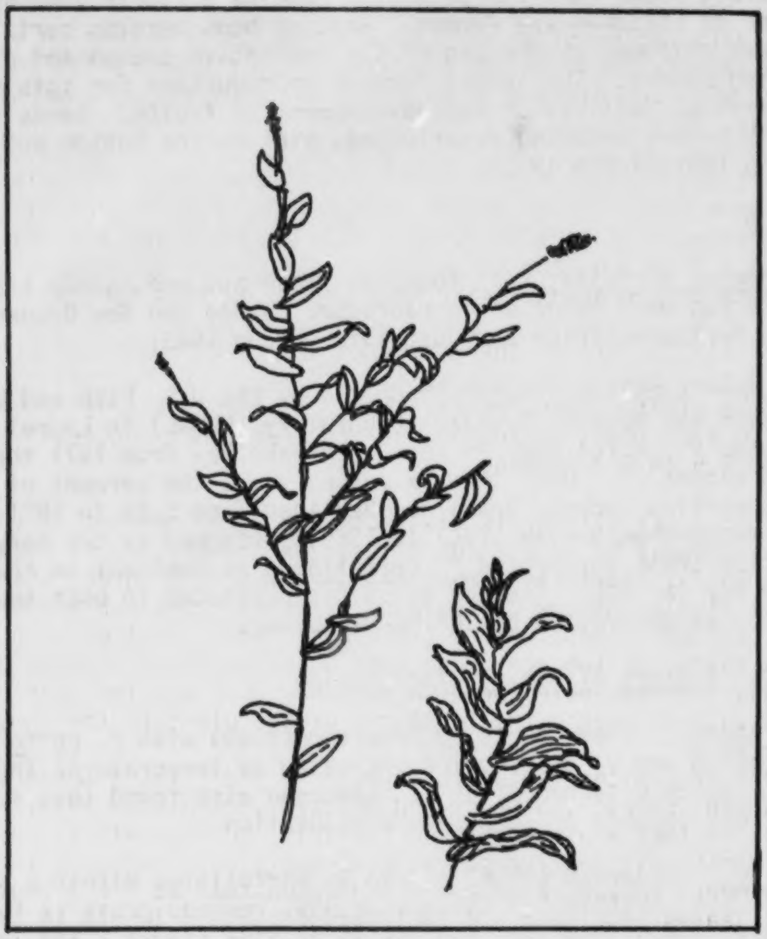
The delicate leaves of this species are flat, scarcely crisped at the margin and have from 7 to 17 nerves (Ogden 1943) (see Figure 1). The upper leaves are ovate, 1 to 3 cm long, while the lower leaves are ovate to lanceolate, 2.5 to 4.5 cm long (Fernald 1970). All of the leaf bases are cordate-clasping which is a characteristic referred to by this species' name.

Fernald (1970) further characterized the species. Stems are slender and straight and the lower stem is simple, becoming more branched toward the upper portion of the plant. Stipules are short and inconspicuous; peduncles are slender, 2 to 6 cm long; spikes are 0.7 to 2 cm long; and fruit is slender, obovoid, and 2.5 to 3.2 mm long.

Potamogeton perfoliatus is distinct from other members of the genus due to its conspicuously heterophyllus foliage yet completely submerged existence (Schulthope 1967). Variation among this species is so great that two plants grown from a single rhizome or isolated branched stems can appear to be separate species (Ogden 1943). Arber (1920)observed a shoot of P. perfoliatus placed in a rainwater tub and found that when the larger leaves decayed after a few months, the new leaves were so much narrower and less perfoliate that it was difficult to relate the two forms of the plant to the same species.

In areas of limited depth, foliage of P. perfoliatus tends to become less brilliant green, shorter, broader and thicker (Hutchinson 1975). A representative series of leaf measurements by Pearsall and Pearsall (1923) indicated that water depths also influence the ratios of leaf length to leaf breadth. Their experiments in Lake Windermere indicated that at a 6 m depth, P. perfoliatus leaves displayed a 4 to 1 ratio (7.0 cm length to 1.7 cm breadth) as compared to a 3 m depth where the leaf ratio was 2 to 1 (3.3 cm length to 1.4 cm breadth).

Factors other than depth operate to alter the morphology of P. perfoliatus. Pearsall and Pearsall (1923) believed that a short, broad-leaf form was characteristic of more calcareous substrata (1600 ppm $Ca(CO_3)_2$ in dry littoral mud). The extreme lanceolate form occurred on much less calcareous sediment (90 ppm $Ca(Co_3)_2$ in dry littoral mud). Pearsall and Hanby (1925) suggested that calcium enhanced the permeability of dividing cells and promoted cell division while potassium promoted cell elongation.



(copied from Hotchkiss 1967)

Figure 1. Redhead grass (Potamogeton perfoliatus)

A distinct anatomical feature of P. perfoliatus is the continued presence of stomata. These structures are functionless because penetration of water to the internal tissues is prevented by a persistent roof of cuticle (Porsch 1905, cited in Sculthorpe 1967).

Reproduction. Modes of propagation for P. perfoliatus include seeds, rootstocks and cuttings (Martin and Uhler 1939). The creeping stem exhibited in this plant is formed by the end-to-end union of the first internodes constituting erect stems (Sculthorpe 1967). During a single growing season, a large number of rhizomes are formed. Resting buds develop serially from the apex of these rhizomes at the end of the vegetative season and produce the next year's spring shoots. The sexual form of reproduction for this species includes flowering, pollination and development of fruits. Seeds float for a short period before becoming waterlogged, sink to the bottom and remain dormant until spring (Hutchinson 1975).

Distribution

Potamogeton perfoliatus is found in fresh and moderately brackish waters. Its presence has been recorded in Labrador, Quebec and New Brunswick and extends to Eurasia, northern Africa and Australia (Ogden 1943).

The summer sampling program conducted by the U.S. Fish and Wildlife Service Migratory Bird and Habitat Research Laboratory (MBHRL) in Laurel, Maryland, has documented P. perfoliatus in the Chesapeake Bay from 1971 through 1976 (see Table 1 and Figure 2). Over the six survey years the percent of sampling stations supporting redhead grass has declined from 5.29 in 1971 to 2.23 in 1976. The Vegetation Survey (1967 to 1969) performed by the Maryland Wildlife Administration (MWA) documented P. perfoliatus as dominant in the Cnoptank River and Eastern Bay in 1968. Redhead grass has persisted in both these areas through 1976, as documented by the MBHRL Survey.

Environmental Factors Affecting Distribution

Temperature. Experiments by Anderson (1969) with P. perfoliatus showed that respiration and O_2 consumption increased as temperatures increased from 25 to 40 C. Death occurred at 45 C. Anderson also found that as redhead grass matured, it was capable of temperature adaptation.

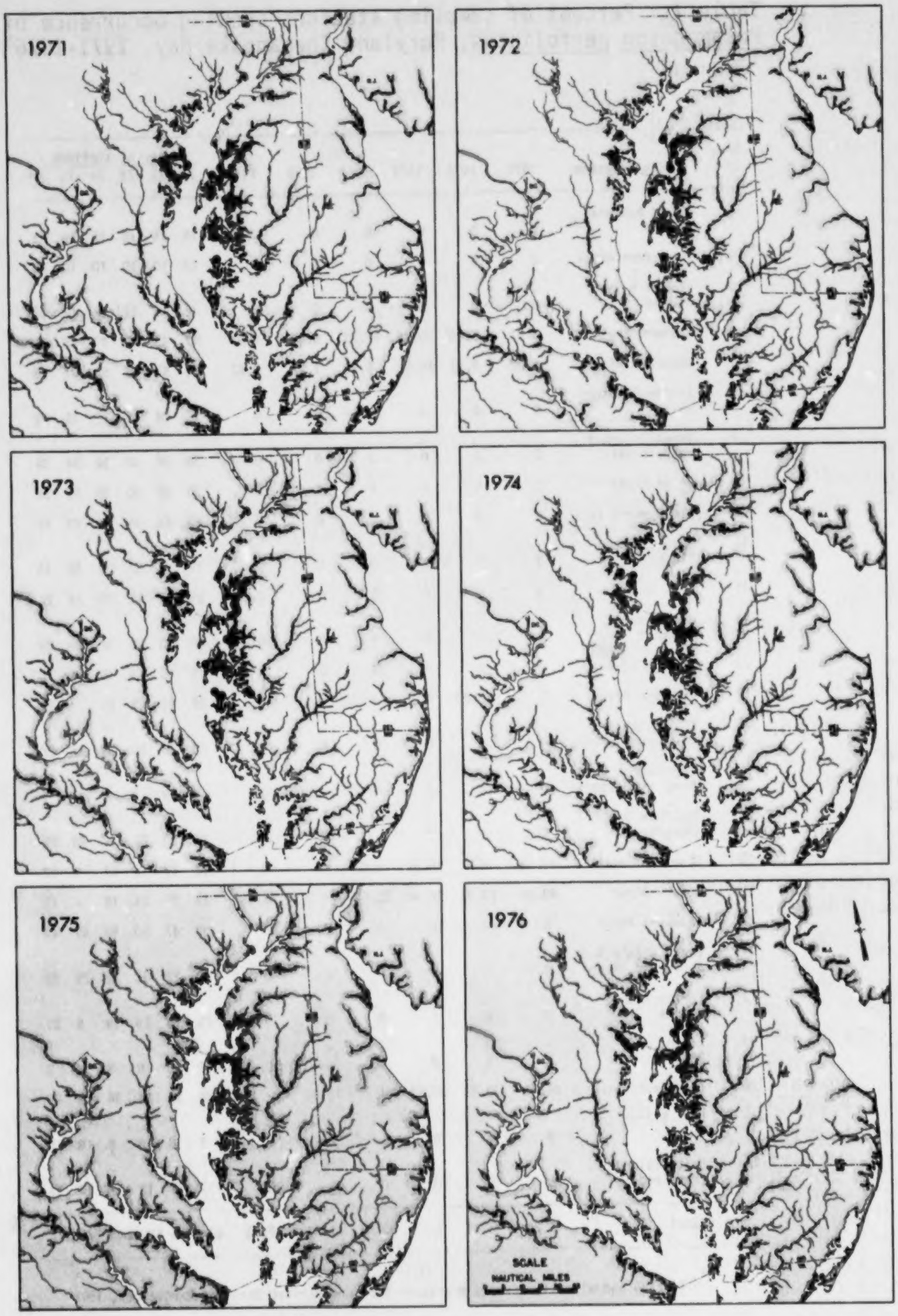
Salinity. Anderson (1969) placed P. perfoliatus within a salinity range from 5 to 25 ppt. Within the Chesapeake Bay, redhead grass is found from the Patapsco River south into the Choptank River (see Figure 2 and Table 1). Salinities in this center portion of the Bay range from about 1.5 to 19 ppt (see Table 81), somewhat lower than Anderson's limits.

Substrate. Pearsall (1920) maintained that within broad limits of depth or light intensity the main limiting factor determining vegetation was the physiocochemical nature of the sediment. Misra (1938) studied in detail the substrate requirements of various submersed plants in English lakes using several sediments types. P. perfoliatus was found to grow best on a mixture of organic material and silt with a minimum carbon to nitrogen ratio, a high capacity to recycle ammonia and a low redox potential. The low redox potential

Table 1. Percent of sampling stations showing occurrence of *Potamogeton perfoliatus*, Maryland Chesapeake Bay, 1971-1976[a]

Area Code	River system	1971	1972	1973	1974	1975	1976	Number of stations					
								71	72	73	74	75	76
1	Elk & Bohemia Rivers	0	0	0	0	0	0	15	16	16	16	16	16
2	Sassafras River	0	0	0	0	0	0	10	10	10	10	10	10
3	Howell & Swan Points	0	0	0	0	0	0	12	6	12	12	12	12
4	Eastern Bay	8.51	6.98	10.64	12.77	6.52	6.67	47	43	47	47	46	45
5	Choptank River	5.00	5.17	10.53	6.90	1.72	3.57	60	58	57	58	57	56
6	Little Choptank River	0	0	0	0	0	0	19	19	19	19	19	19
7	James Island & Honga River	0	0	0	0	0	0	34	34	34	34	34	34
8	Honga River	0	0	0	0	0	0	30	30	30	30	29	29
9	Bloodsworth Is.	0	0	0	0	0	0	40	44	46	43	43	45
10	Susquehanna Flats	0	0	0	0	0	0	27	37	37	37	36	35
11	Fishing Bay	0	0	0	0	0	0	25	25	25	25	24	25
12	Nanticoke & Wicomico Rivers	0	0	0	0	0	0	30	30	30	30	30	30
13	Manokin River	0	0	0	0	0	0	15	15	15	15	14	15
14	Patapsco River	0	5.00	4.76	4.76	-	0	21	20	21	21	-	21
15	Big & Little Annemessex Rivers	0	0	0	0	0	0	20	20	20	19	18	20
16	Gunpowder & Bush River Headwaters	0	0	0	0	-	0	9	8	7	9	-	9
17	Pocomoke Sound (Maryland)	0	0	0	-	0	0	22	20	21	-	20	22
18	Magothy River	33.33	-	16.67	16.67	-	0	12	12	12	12	-	12
19	Severn River	40.00	13.33	20.00	20.00	-	23.08	15	15	15	15	-	13
20	Patuxent River	0	0	0	0	0	0	50	47	50	50	47	49
21	Back, Middle & Gunpowder Rivers	0	0	0	0	0	0	22	22	22	22	22	22
22	Curtis & Cove Points	0	0	0	0	0	0	20	19	19	19	6	21
23	South, West & Rhode Rivers	0	0	0	0	0	0	8	10	10	8	8	8
24	Chester River	44.44	33.33	26.47	14.71	16.67	17.14	36	36	34	34	36	35
25	Love & Kent Points	0	0	0	0	0	0	8	8	8	8	8	8
26	Smith Island (Maryland)	0	0	0	0	0	0	17	11	12	17	17	17
	Total	5.29	3.41	4.13	3.44	1.81	2.23	624	615	629	610	553	628

[a] U.S. Fish and Wildlife Service Migratory Bird and Habitat Research Laboratory, 1976


1971
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1976
SCALE
NAUTICAL MILES

Figure 2. Distribution of Potamogeton perfoliatus, Maryland Chesapeake Bay, 1971-1976.

suggested an abundance of bacterial action and a high content of exchangeable cations. Thus it was determined that moderately organic muds fairly rich in nitrogen and exchangeable calcium were more suitable than highly organic muds.

Light, Depth and Turbidity. Potamogeton perfoliatus is usually found in either still or standing water ranging from 0.6 to 1.5 m in depth (Martin and Uhler 1939). Felfoldy (1960) found that a maximum rate of photosynthesis was attained at a depth of 2 m on a day where the light intensity was about 1.1 g cal/cm^2.

Nutrient Responses. Potamogeton perfoliatus is one of the two most common species observed by Lohammar (1965) in Swedish lakes. This macrophyte occurred throughout the greater part of the concentration range of combined nitrogen, total potassium, phosphorus and calcium in waters of these lakes. Under conditions where calcium was limited, foliar growth was radically altered.

Susceptibility. Generally, Potamogeton is not highly susceptible to heavy metals. Petkova and Lubyanov (1969 cited in Hutchinson 1975) found marked accumulation of vanadium consisting of 1300 ppm in the plant ash. Potamogeton perfoliatus did not respond to 2,4-D(PGBEE) applied in granular form at 44.4 kg/ha a.i. (Lawrence and Hollingsworth 1969). However, treatments of simazine, monuron and silvex (PGBEE) applied at the same rates as mentioned for 2,4-D, controlled redhead grass. Endothall applied at 90.7 kg/ha a.i. was found to completely control plant growth, while an application of 14.3 kg/ha a.i. affected partial control.

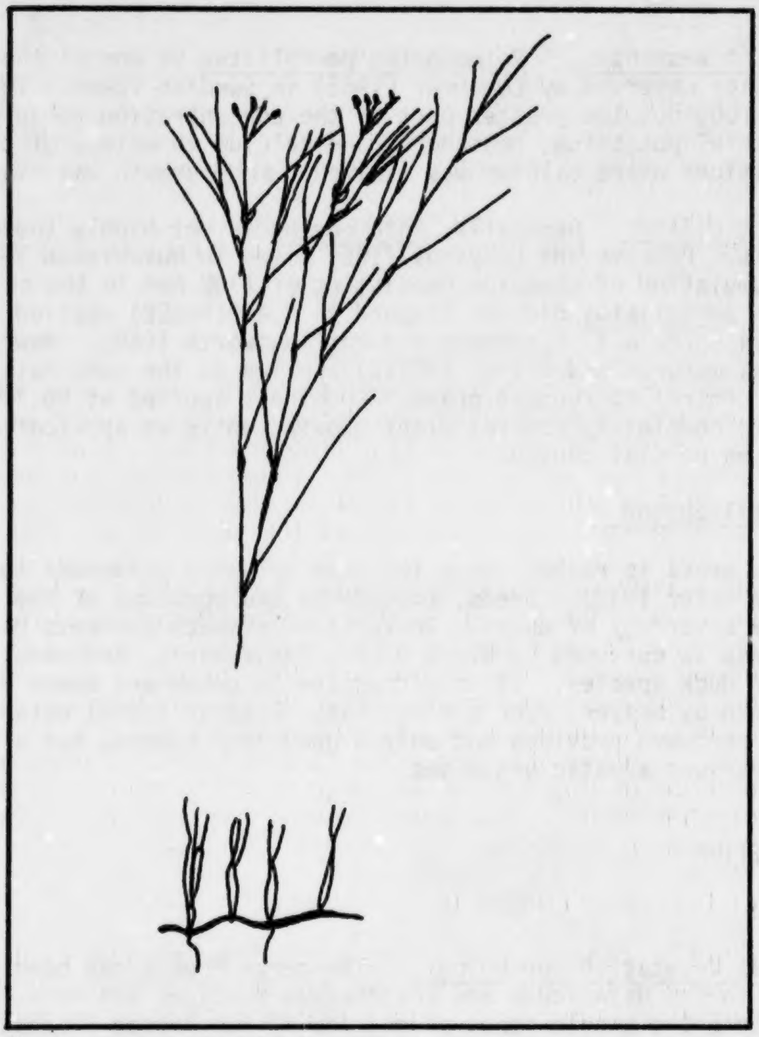
Consumer Utilization

Redhead grass is ranked among the more valuable pondweeds to waterfowl (Martin and Uhler 1939). Seeds, rootstocks and portions of the stem are consumed by a variety of ducks. Analysis cf stomach contents has indicated that redhead grass is consumed by Black Ducks, Canvasbacks, Redheads, Ringnecks, among other duck species. It is attractive to geese and swans and often heavily eaten by beaver, deer and muskrat. Fassett (1960) noted that this species of pondweed provides not only a good food source, but also protective cover for various aquatic organisms.

RUPPIA MARITIMA

Biology

General Vegetative Morphology. The genus Ruppia has been variously classified in the Najadaceae and Zosteraceae families but more recently has been separated into the single genus of the family Ruppiaceae (Takhtajan 1969). Ruppia maritima is a highly variable, slender, branching perennial herb with linear or filiform opposite leaves 2 to 20 cm long and about 1 to 2 mm broad (Welsh 1974; Fasset 1966; Weldon et al. 1969)(see Figure 3). Stems are generally terete (Welsh 1974) and may be up to 3 m long (Weldon et al. 1969). In shallow waters, R. maritima plants have been observed so short as to appear like a carpet of leaves 3 to 10 cm tall without stems (Weldon et al. 1969).



(copied from Hotchkiss 1976)

Figure 3. Widgeongrass (Ruppia maritima)

Commonly known as widgeongrass, Ruppia produces only submerged leaves and is not capable of survival under direct sunlight (McCann 1945). The species has an extensive root system (Weldon et al. 1969) composed of much branched creeping rhizomes (McCann 1945) and no tubers (Hotchkiss 1967; Radford et al. 1964). Flowers are perfect, small and borne on axillary stems (Cook et al. 1974). Up to the time of flowering, the inflorescence is enclosed in a sheath formed by the two uppermost leaves (Rendle 1930).

Reproduction. Ruppia maritima reproduces both vegetatively and sexually. Vegetative propagation occurs primarily through the rhizomes (U.S. Department of Interior 1944). Sexual reproduction involves the elongation of the peduncle upwards to the air water interface (Rendle 1930). Once at the surface, the curved, tubular pollen (Rendle 1930) is released and floats on the surface until it contacts the floating stigmas (Arber 1920). McCann's (1945) personal observations concerning the Ruppia pollination mechanisms described pollination as occuring below the water surface. The two sets of anthers, one above and one below the female cluster, shed their pollen slowly and the pollen drifts upwards and adheres to the stigmatic canopy. Fertilization takes place when the pollen drifts around to the stigma.

Distribution

Widgeongrass inhabits a wide range of shallow, brackish pools, rivers and estuaries along the Atlantic, Gulf and Pacific Coasts (Martin et al. 1951; Radford et al. 1964). Ruppia also flourishes in alkaline lakes, ponds and streams and in shallow, saline ponds and river deltas of the Great Salt Lake region (Ungar 1974). Widgeongrass is not limited to brackish or salt water, but also occurs in fresh portions of estuaries (Chrysler et al. 1910).

The MBHRL summer survey has documented Ruppia maritima from the Back, Middle and Gunpowder Rivers south to the Maryland/Virginia state line from 1971 to the present (see Table 2 and Figure 4). Data from the six years indicates a slight downward trend in vegetation from 1971 to 1976. However, the study shows a strong positive trend from 1975 to 1976, almost back up to the 1971 level. Data for 1972 indicate that Ruppia probably was not drastically affected by the salinity decreases due to tropical storm Agnes. The decrease in percentage occurrence in 1975 may be due in part to the fact that the Severn River was not sampled that year after showing consistently high percentages of widgeongrass in previous years.

Environmental Factors Affecting Distribution

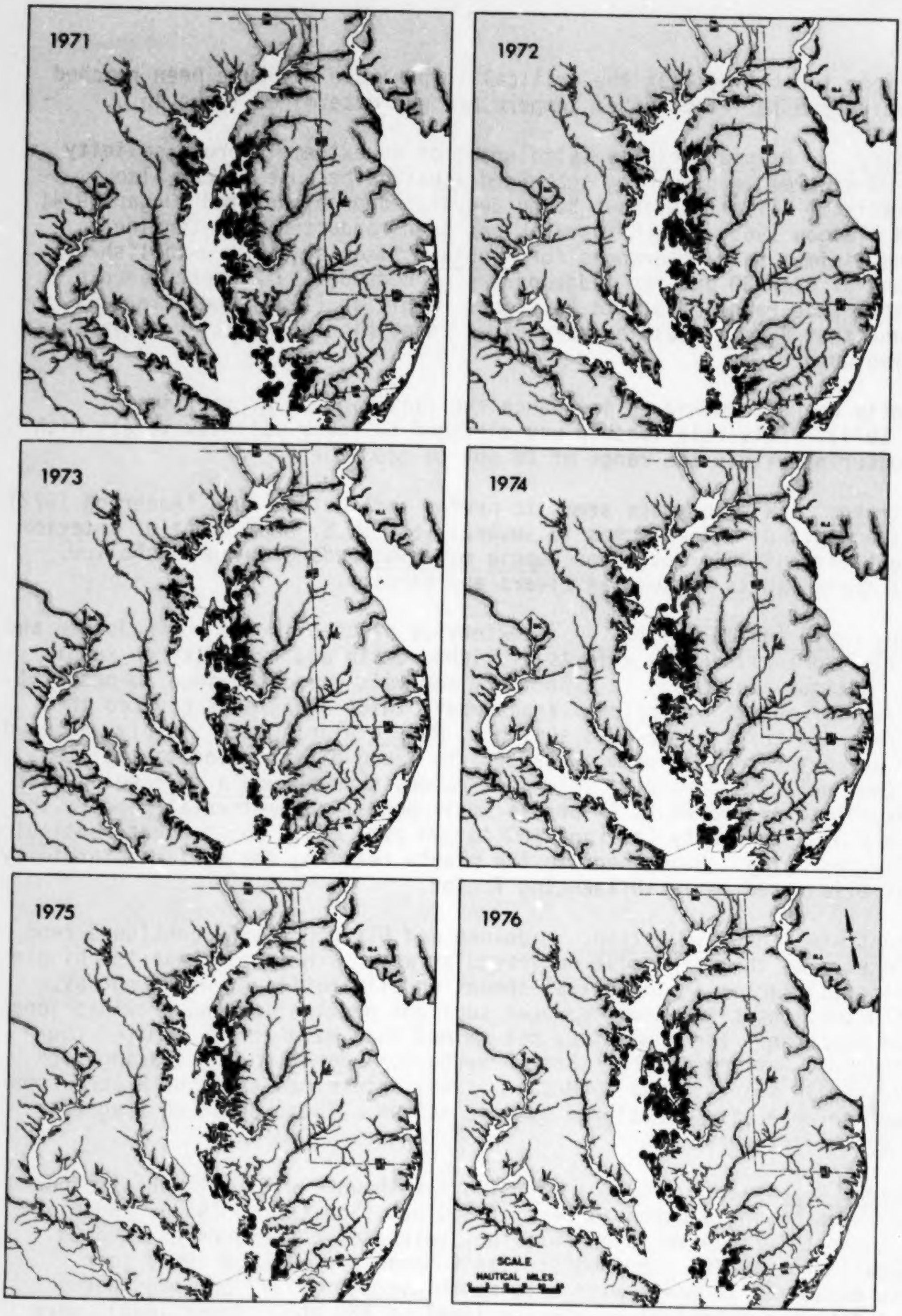
Temperature. Pond studies by Joanen and Glasgow (1965) showed that R. maritima appeared to have two growing seasons occurring within the temperature range of 18 to 30 C. Growth apparently ceased outside this range; however, some fruiting and flowering were observed at temperatures higher than 30 C.

Anderson (1969) conducted experiments in the Patuxent River near the effluent of an electrical generating station. Anderson concluded that new growth from rhizomes, seed germination and flowering all had critical temperature ranges. There was a significant reduction in aerial coverage of plants near the

Table 2. Percent of sampling stations showing occurrence of *Ruppia maritima*, Maryland Chesapeake Bay, 1971-1976[a]

Area Code	River system	1971	1972	1973	1974	1975	1976	Number of stations					
								71	72	73	74	75	76
1	Elk & Bohemia Rivers	0	0	0	0	0	0	15	16	16	16	16	16
2	Sassafras River	0	0	0	0	0	0	10	10	10	10	10	10
3	Howell & Swan Points	0	0	0	0	0	0	12	6	12	12	12	12
4	Eastern Bay	23.40	30.23	23.40	34.04	17.39	37.78	47	43	47	47	46	45
5	Choptank River	23.33	31.03	10.53	24.14	1.72	39.29	60	58	57	58	57	56
6	Little Choptank River	15.79	10.53	0	0	0	15.79	19	19	19	19	19	19
7	James Island & Honga River	17.65	8.82	2.94	5.88	5.88	8.82	34	34	34	34	34	34
8	Honga River	30.00	23.33	10.00	16.67	10.34	13.79	30	30	30	30	29	29
9	Bloodsworth Is.	20.00	2.27	8.70	4.65	4.65	2.22	40	44	46	43	43	45
10	Susquehanna Flats	0	0	0	0	0	0	27	37	37	37	36	35
11	Fishing Bay	8.00	0	0	0	0	0	25	25	25	25	24	25
12	Nanticoke & Wicomico Rivers	0	0	0	0	0	0	30	30	30	30	30	30
13	Manokin River	20.00	13.33	6.67	13.33	7.14	6.67	15	15	15	15	14	15
14	Patapsco River	0	0	0	0	-	0	21	20	21	21	-	21
15	Big & Little Annemessex Rivers	45.00	15.00	25.00	31.58	16.67	25.00	20	20	20	19	18	20
16	Gunpowder & Bush River Headwaters	0	0	0	0	-	0	9	8	7	9	-	9
17	Pocomoke Sound (Maryland)	9.09	0	0	-	5.00	4.55	22	20	21	-	20	22
18	Magothy River	8.33	0	0	8.33	-	0	12	12	12	12	-	12
19	Severn River	33.33	20.00	13.33	26.67	-	15.38	15	15	15	15	-	13
20	Patuxent River	-	4.26	-	2.00	0	2.04	50	47	50	50	47	49
21	Back, Middle & Gunpowder Rivers	4.55	0	0	0	0	0	22	22	22	22	22	22
22	Curtis & Cove Points	0	0	0	0	0	0	20	19	19	19	6	21
23	South, West & Rhode Rivers	0	0	0	0	0	12.50	8	10	10	8	8	8
24	Chester River	27.78	11.11	8.82	2.94	11.11	14.29	36	36	34	34	36	35
25	Love & Kent Points	0	0	0	12.50	0	0	8	8	8	8	8	8
26	Smith Island (Maryland)	47.06	27.27	16.67	29.41	5.56	35.29	17	11	12	17	17	17
	Total	14.74	9.92	6.04	9.84	4.69	11.46	624	615	629	610	553	628

[a] U.S. Fish and Wildlife Service Migratory Bird and Habitat Research Laboratory, 1976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1972
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1975
1976
SCALE
NAUTICAL MILES

Figure 4. Distribution of Ruppia maritima, Maryland Chesapeake Bay, 1971-1976

power station, indicating that the critical temperature may have been reached for vegetative growth. The lethal temperature was determined to be 45 C.

Salinity. Ruppia maritima is tolerant of an extremely broad salinity range and considered unique among submerged aquatics because it can also tolerate variable salinity without being dehydrated or hydrolized (Ungar 1974; Joanen and Glasgow 1965). Much research has been conducted to determine maximum and minimum salt tolerances for Ruppia. Steenis (1970) established a tension zone of over 30 ppt for widgeongrass. Anderson (1972) determined a salinity tolerance range of 5.0 to 40.0 ppt. McMillan (1974) and Osterhaut (1906) found that Ruppia could be maintained indefinitely in either tapwater or distilled water.

Salinity was determined to influence the flowering of R. maritima (McMillan 1974). This relationship was confined to lower salinity levels with seed set occuring within the range of 28 ppt to tapwater.

Substrate. Widgeongrass seems to prefer soft bottom muds (Anderson 1972) or sand at a depth of a few inches to several feet (U.S. Department of Interior 1944). Zenkevitch (1963) described Ruppia maritima beds growing on shallow sand shell gravel soils in Russian rivers and streams.

Light, Depth and Turbidity. In extensive studies in Louisiana, Joanen and Glasgow (1965) determined the effects of light, depth and turbidity on Ruppia maritima. Optimum production in laboratory analysis was determined to occur at a water level of 60 cm. In natural areas where thick algal mats covered the water surface and totally excluded sunlight, Ruppia plants were visibly weakened. Turbidity was determined to be most harmful to young plants prior to the stems reaching the surface. Turbidity tolerance ranges showed Ruppia to develop best at a level not to exceed 25 to 35 ppm in small ponds. Experimental ponds with persistently high turbidity (average 100 to 108 ppm) could not support continuing growth of R. maritima. Subsequent to the plants reaching the surface, turbidity was almost eliminated as an influencing factor.

Current Wind and Wave Action. Joanen and Glasgow (1965) concluded from their pond studies that excessive or irregular water fluctuation was the single most detrimental factor in the establishment or elimination of widgeongrass. Extreme fluctuations alternately reduced sunlight penetration and provided long periods of exposure. The same study determined that wave action limited the growth of Ruppia maritima either through mechanical damage or by causing high turbidity. Field observations by McCann (1945) determined that the branch tops at the surface were often fractured by wave action. These detached fragments were not capable of survival.

Nutrient Responses. Ruppia maritima has shown a high tolerance to boron in saline lakes having a boron content of 850 ppm (Sculthorpe 1967). Ruppia has also shown tolerance for high chlorine, sulphate, phosphorus and sodium levels (Sculthorpe 1967). In pond studies (Joanen and Glasgow 1965) poor growth was exhibited in ponds with a 10-month average of 177 ppm phosphorus while best growth occurred at an average level of 120 ppm. Toxic levels were not established. The study further determined "...no correlation between plant growth and calcium or that this element at any time inhibited plant growth."(p.13).

Soil pH was not determined to be a limiting factor for Ruppia maritima growth; plants grew vigorously in both high and low pH soils.

Susceptibility. Ruppia maritima has not been utilized extensively in experiments to determine relative toxicity levels to various herbicides. A possible explanation for this lack of information may be the recognized importance of Ruppia as waterfowl food. Further, the majority of the herbicide toxicity experiments have been conducted in a fresh water media. Ruppia is not often documented as a fresh water inhabitant.

Table 3 lists those herbicides documented by the U.S. Department of Agriculture for controlling Ruppia.

Table 3. Effects of certain aquatic herbicides on Ruppia maritima[a]

Herbicide	Application (ppm)	Control
Sodium arsenite	1	Promising
Endothall, DDS	1.5 2	None 10 %
Acrolein	5.6 to 6.8	Good

[a] Lawrence and Hollingsworth 1969.

CIBA-GEIGY (personal communication 1977) has not included Ruppia maritima on their label for Aquazine (simazine). Their only information concerning Ruppia refers to complete control with Aquazine at 1.0 ppm in a North Carolina lake. Similar control could probably be expected with an atrazine concentration of 0.25 to 0.50 ppm.

Productivity

There is a scarcity of information available concerning productivity of Ruppia maritima in the Chesapeake Bay. Anderson (1966) was able to compare primary productivity of Ruppia between the Patuxent River and Chincoteague Bay He found Ruppia to be considerably more productive in the Patuxent River (800 g/m^2 average mean dry weight)than in Chincoteague Bay (250 g/m^2 average mean dry weight). The height of the growing season for Ruppia in Chincoteague Bay seemed to occur from about the middle of June to the end of July. Growth did not necessarily cease when the elongating stems reach the surface, so that leaf extension and expansion remained possible.

Consumer Utilization

Ruppia maritima is rivaled only by Zostera marina in importance to waterfowl (Bourn 1935). Widgeongrass serves as food for numerous species of ducks, coots, geese, grebes, swans, marsh and shore birds and game birds of the Atlantic, Pacific and Gulf Coasts (Sculthorpe 1967; Martin and Uhler 1939). Studies of the upper Chesapeake Bay area over the period 1959 to 1968 (Rawls in press) showed Ruppia Maritima to have the highest frequency-of-use rate for the waterfowl and plant species studied. Out of 1,179 geese, dabbling duck, diving duck and Meganser gizzards utilized during the survey, 363 birds (30.79 percent) contained widgeongrass plant parts. Volumetric analysis rated Ruppia third after Zea mays and Potamogeton perfoliatus.

A study performed in the 1930s (Cottam 1934c) on the gullets of 3,499 North American inland plant feeding ducks, included Ruppia Maritima in the list of aquatic plants having the greatest food value to Nyroca and ruddy ducks. Waterfowl were found to have eaten all parts of the Ruppia plants. Pond studies in Louisiana (Joanen and Glasgow 1965) showed waterfowl utilization of Ruppia so heavy as to consume entire plant stands.

Ruppia beds are also used as nursery grounds (Sculthorpe 1967) and as a fish spawning medium and cover for marine organisms (Kerwin 1975b). Sculthorpe (1967) also considered Ruppia stands of great importance to fish as a food and shade source.

Economic Uses and Problems

Impoundments. Baldwin (1967), in studying the use of impoundments for the management of waterfowl, concluded that when considering all types of impoundments, Ruppia maritima ponds probably provided more waterfowl food with the least amount of management. In creating impoundments, Baldwin suggested that the following conditions would be delterious to Ruppia propagation: stained, acidic water; excessive upland runoff; surplus fresh water; silt and detritus; and organically stained water. The relatively high salinity tolerance of Ruppia allows for the prevention of rapid changes caused by natural plant succession and invasion by fresh water pests such as cattail (Typha spp.) (Ball 1965).

MYRIOPHYLLUM SPICATUM[1]

Biology

General Vegetative Morphology. Eurasian watermilfoil (Myriophyllum spicatum) belongs to the order Hippuridales, family Haloragaceae (Takhtajan 1969). The plant was first decribed by Linnaeus in 1953 (Anderson et al. 1965). Disagreement persists in the literature as to the proper taxonomic relationship to a very similarly appearing species, Myriophyllum exalbescens. M. spicatum is closely related to M. exalbescens, but the latter species is found mostly in glaciated areas (Steenis and Stotts 1961). Beaven (1962), in his summary report included a suggestion by Dr. Paul Springer that new reports of M. spicatum in the Northern United States may have been the results of misidentification of M. exalbescens. Love (1961) maintained that differences between the two species are so pronounced that they can confidently be considered to be separate species. The Terminology Committee of the Weed Society of America (Klingman 1962) has given approval to common names of Eurasian watermilfoil for M. spicatum and to northern watermilfoil for M. exalbescens.

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[1] Special contribution from Charles K. Rawls, University of Maryland Center for Environmental and Estuarine Studies, Chesapeake Biological Laboratory.

Myriophyllum spicatum is an aquatic perennial with verticilliate submerged leaves which are pinnatified (Southwick 1972) (see Figure 5). Springer (1959) described milfoil as generally having leaves with 14 to 21 pairs of upward curving, threadlike divisions and stems that are greenish and remain this color when dry. Springer et al. (1961) compared milfoil leaves to weathered, watersoaked feathers. M. spicatum is much branching, variable in growth under various conditions and with leaflets near the flowering tips more numerous (usually 13 or more) than in other milfoil species (Stotts 1961). The leaflets are about 1.8 cm long (Haven 1961). The flowers are axillary and monoecious.

Reproduction. Patten (1956) studied the flowering and reproductive cycles of the perennial M. spicatum in the shallow, alkaline, eutrophic New Jersey Lake Musconetcong. Flowering began in late May and continued until early October with the earliest flowering occurring in the shallower lake depths.

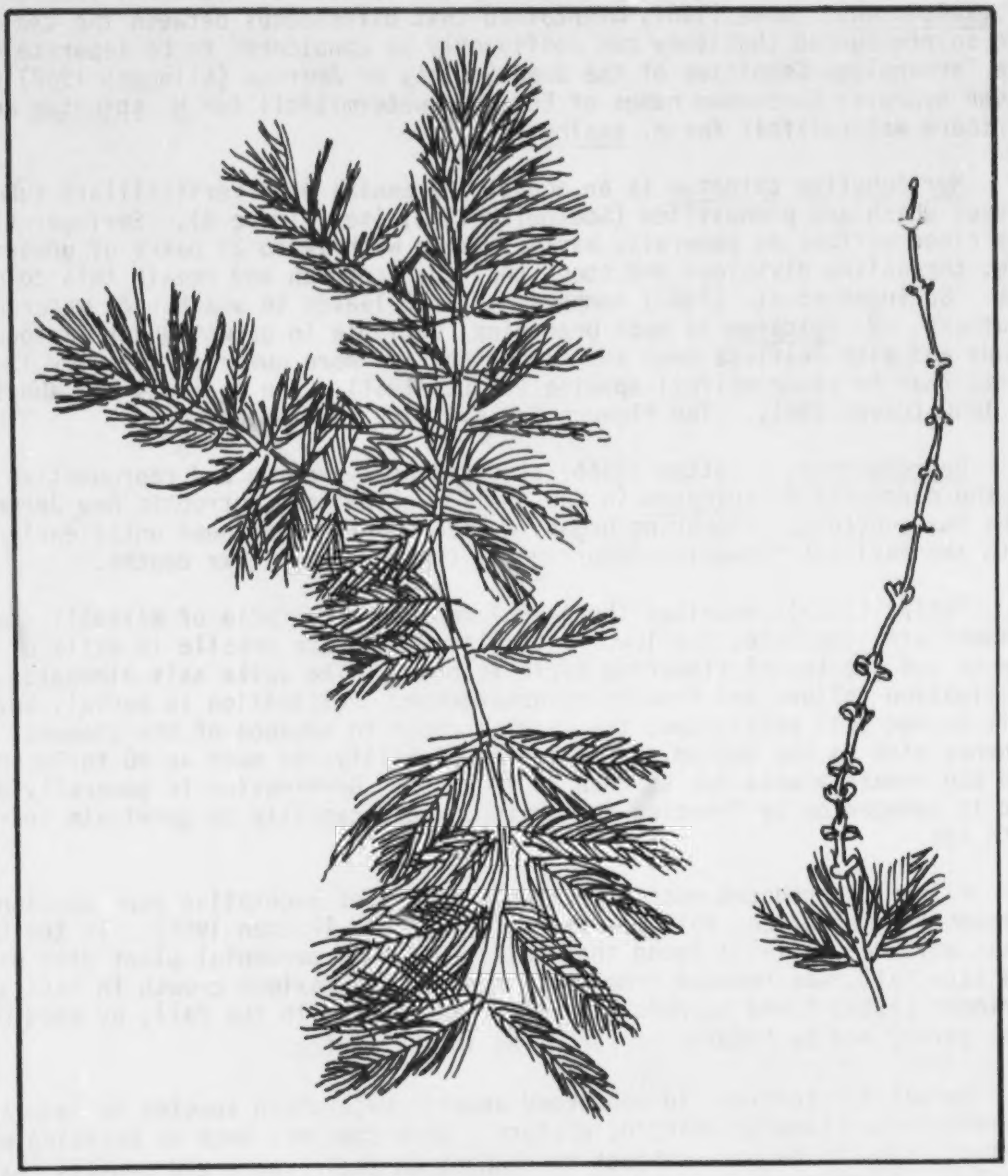
Patten (1955) described the sexual reproductive cycle of milfoil. Upper flowers are staminate, the lower pistillate, both are sessile in axils of strong bracts and the sexual flowering cycle is brief. The spike axis elongates, then pollination follows and finally resubmergence. Pollination is aerial, but milfoil is not self pollinated; the stigmas ripen in advance of the stamens. The achenes sink to the bottom, exhibit high viability (as much as 80 to 90 percent) and can remain viable for as much as 10 years. Germination is generally delayed, but is speeded up by freezing and drying. Seed capacity to germinate increases with age.

Milfoil reproduces most effectively by way of vegetative reproduction through fragmentation, rhizomes and axillary buds (Patten 1956). In the Chesapeake Bay, Haven (1961) found that foliage of this perennial plant died back in the late fall, was renewed from roots and reached maximum growth in late summer. Springer (1959) found reproduction by fragmentation in the fall, by seeding (no time given) and by budding or sprouting in the spring.

Annual fluctuations in submerged aquatic vegetation species is natural and presents a continually changing picture. Some species, such as Eurasian watermilfoil, seem to be more subject to changes in their range and density than other species. In most areas, milfoil has two growth-surge periods during the growing season. To keep abreast of such changes, two or three surveys of vegetation beds should be made annually (Rawls 1971a).

Distribution

There is some controversy as to where milfoil was originally introduced on the East Coast of the United States. Beaven (1962) noted that the plant originally may have been introducted between 1880 and 1890 when ships coming from Europe



(copied from Hotchkiss 1967)

Figure 5. Eurasian watermilfoil (Myriophyllum spicatum)

dumped their ballast. Others (Bayley et al. in press; Steenis 1970; Rawls 1964, 1975; Steenis et al. 1962; Springer et al. 1961; Elser 1966; Steenis et al. 1967) have traced the early history of milfoil back to various dates ranging from 1895 to 1902.

From the turn of the century until the 1950s, milfoil was found sporadically from the C & D Canal in Baltimore south into North Carolina. However, according to Virginia Institute of Marine Science (VIMS), Chesapeake Bay Laboratory (CBL) and Migratory Bird and Habitat Research Laboratory (MBHRL) personnel, rapid spread did not begin until around 1954. By 1959 live fragments of milfoil could be found floating all over the Bay (Steenis et al. 1962). Bay acreage infested by milfoil increased from 20,200 ha in 1960 to 40,500 ha in 1961. An example of the rapidity with which milfoil infested the Bay was seen on the Susquehanna Flats where a survey showed that the percentage of established transect stations with milfoil was 0 in 1957, 1 percent in 1958, 47 percent in 1959, 94 percent in 1960, 71 percent in 1961 and 88 percent during another 1961 sampling of the same stations (Steenis et al. 1962).

As dramatically as milfoil infested the Bay from about 1959 to 1965, it declined just as rapidly. However, the decline was not uniform within the entire Bay. From 1959 to 1960, there was a decrease in milfoil infested areas in the Lower Machodoc Creek, Virginia, for no known reason (Stotts 1961). Rawls (1968) described changes in one Potomac River tributary, the Wicomico River. Milfoil was classed as abundant throughout the entire Wicomico River in 1964 and 1965. In 1966 it was abundant only in the upper portion of the river above Mills Point and by 1967 was limited to a small acreage near Allan's Fresh at the head of the river. However, in 1968, there were 80 ha where there had been 5 ha in 1967, and along the western shore between the power line river crossing and Chaptico Bay, rooted milfoil beds covered an estimated 300 to 400 ha in a continuous stretch. The previous year, for the entire Wicomico, the estimate was less than 4 ha of milfoil.

Munro (1976b) discussed the decrease in Chesapeake Bay aquatic vegetation based on the 1971 to 1976 MBHRL survey (see Figure 6). Milfoil declined over the six-year survey from 3.85 percent vegetated stations in 1971 to 0.96 percent vegetated stations in 1976. In attempting to characterize the spectacular invasion of the Chesapeake Bay by milfoil, Bayley et al. (in press) described the milfoil spread on the Susquehanna Flats as a "wave phenomenon" from an epicenter, spreading from the most optimal to the least optimal areas. At first milfoil was co-dominant with *Najas* spp. and *Vallisneria americana*. Bayley et al.(in press) further stated that milfoil stands tended to become homogenic and discrete within three years, but later were structured to include several species of SAV at various densities. In time the milfoil epidemic receded and a new equilibrium was established. In theory an area is most susceptible to an epidemic when the natural ecosystems are disturbed, particularly by man.

The present (summer, 1977) status of Eurasian milfoil in the Chesapeake Bay is uncertain. The Susquehanna Flats Survey and the MBHRL Survey were conducted over July and August. Unless these surveys uncover drastic changes over 1976 populations, there will again be remnant stands and pockets of milfoil in scattered areas of the upper Chesapeake Bay and its tributaries.

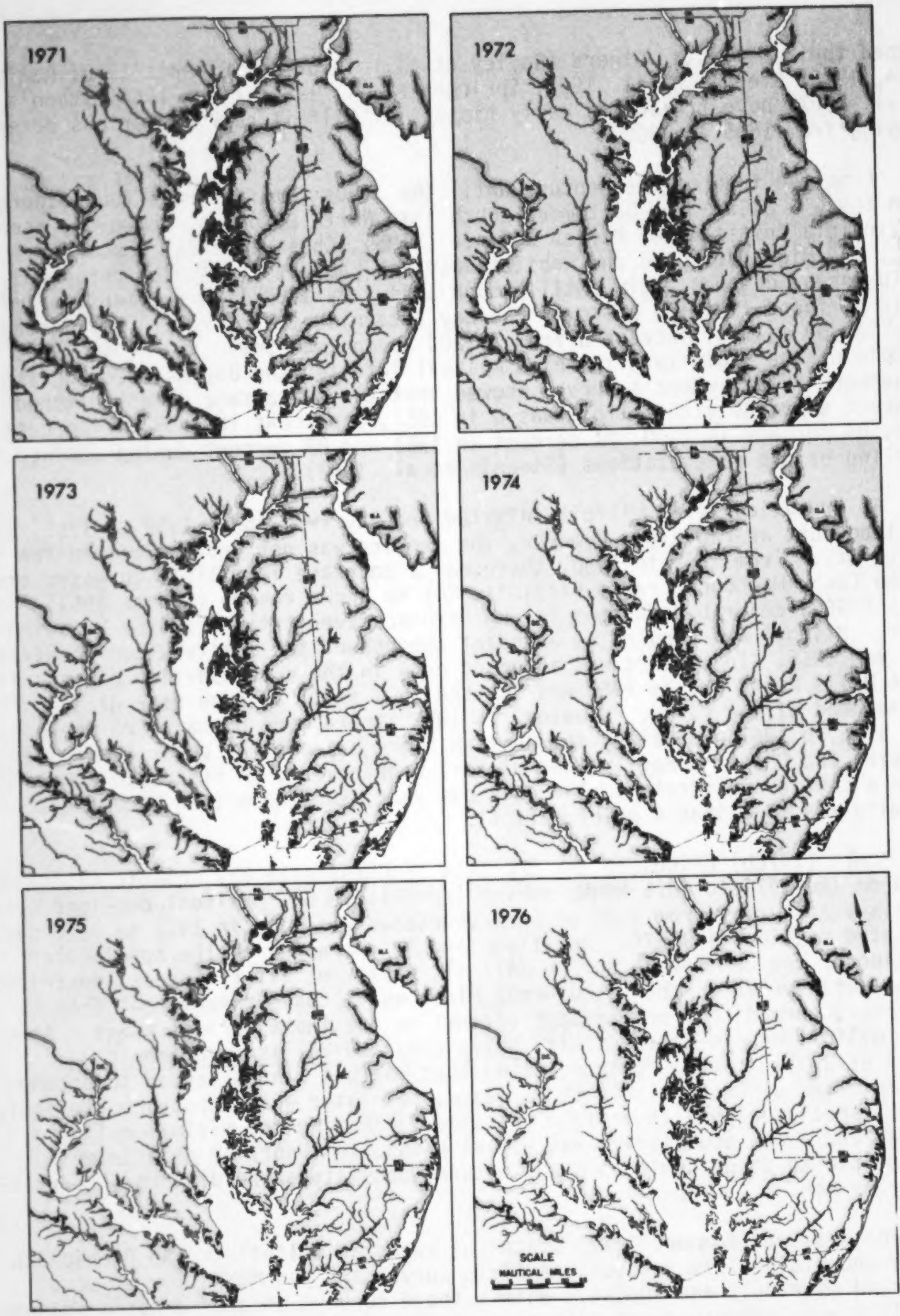

1971
1972
1973
1974
1975
1976
SCALE
NAUTICAL MILES

Figure 6. Distribution of Myriophyllum spicatum, Maryland Chesapeake Bay, 1971-1976

Outside the Chesapeake Bay area, milfoil has also infested many lakes in New York, New Jersey and Tennessee (Springer et al. 1961; Stotts 1961). Eurasian watermilfoil is still a problem in the Kawartha Lake region of Ontario, Canada (Steenis 1976). It also has become a nuisance in such areas of British Columbia as Osoyoos and Kalamalka in Skaha and Okanagan Lakes (Newroth 1977). As one would suspect from the name, it is native to Europe and Asia (Anonymous 1976), and is widespread in Europe, Asia and parts of Africa (Springer 1959). Eurasian watermilfoil is the chief submersed aquatic vegetation species in company with redhead grass (*Potamogeton perfoliatus*) and sago pondweed (*Potamogeton pectinatus*) in the Kiliyskaya Delta of the Russian Danube (Klokov Zimbalevskaya 1974). Milfoil is also known to be in at least the middle course of the Saale River of East Germany (Krausch 1976).

Environmental Factors Affecting Distribution

Temperature. Anderson (1964) and Anderson et al. (1965) found milfoil growing in temperatures ranging from 0.1 to 30 C in the freshwater Twin Ponds. Vigorously growing milfoil was even found under 25 cm of ice.

Titus et al. (1975) described temperature as a primary environmental forcing function. In a production model, Titus et al. determined that a 10 C increase stimulated a 10 percent increase in the peak standing crop.

In the reactor cooling reservior Par Pond, South Carolina, the distribution of aquatic macrophytes was studied by Grace and Tilly (1976). The standing crop of milfoil as measured by biomass, was most abundant at the warm station (average maximum temperature was 23.8 C), was twice as high as the cold station (maximum temperature was 2.2 C) and three times as high as the hot station (maximum temperature was 26.3 C).

Salinity. *Myriophyllum spicatum* customarily inhabits fresh to brackish waters. Under natural conditions, milfoil can be found in salinities ranging from 0 to 20 ppt (Rawls 1964). At 20 ppt, Beaven (1960) found milfoil alive during the summer, but plants exhibited little growth. Only the foliage remaining at the growing tips was alive. From observations in the Chesapeake Bay, Davis et al. (1974) commented that milfoil disappeared as salinities increased to 13 or 14 ppt.

In laboratory studies, Boyer (1960*b*) found milfoil to grow well in water from 0 to 5 ppt. Springer (1959) determined that milfoil rooted best at 3.5 ppt. As salinities were increased, milfoil growth decreased. Boyer (1960) proved this experimentally; inhibition started at 10 ppt and became severe from 15 to 20 ppt. Haller et al. (1974) in similar experiments, found some toxicity to occur in salinities from 10 to 13.3 ppt. At 16.6 ppt, plants died.

Leaf area, stem diameter and extent of flowering was found to decrease as salinity increased, while flowering was delayed or non-existent with sufficient salinity increases. When salinities increased above 15 ppt, stems broke near the bottom and large floating mats of partially decayed plant material formed (Anderson 1964).

Biological stress induced by severe drought in 1933 to 1934 reduced most vegetation in the lower Potomac River to remnant stands. Presumably this was due to salt water intrusion which moved as far up the Potomac River as Mt. Vernon.

Alkalinity, pH, CO_2, O_2. Milfoil is generally absent in more acid waters while the alkalinity of brackish waters favor growth (Steenis and Stotts 1961). Eurasian watermilfoil has an affinity for alkaline waters (Patten 1956) and can precipitate encrusting marl under highly calcareous conditions (Nichols 1975). On August 1, 1962, near the mouth of the South River, Southwick (1972) obtained pH readings from 7.2 to 8.9; free CO_2=0.00, phenolthalein alkalinity = 7.25 to 21.0. The readings were made as water temperatures rose from 25 C to 29 C. Based on measurements obtained over a 13-hour tidal cycle in the Wicomico River (Potomac River estuary) Anderson et al. (1965) concluded that a pH range from 5.8 to 9.7 and a low concentration of CO_2 were not limiting factors. In field studies, Rawls (1977) reported a pH range of 7.9 to 9.3 between June 24 and October 16, in Chaptico Bay. Oxygen during the same period was measured from 5.6 to 9.9 ppm at the surface and 6.64 to 7.66 ppm at the bottom in milfoil infested waters. Southwick (1972) stated that milfoil could survive in oxygen depleted waters (no levels given).

Substrate. Studies by Patten (1956) in Lake Musconetcong, New Jersey, correlated milfoil density to substrate particle size. Maximum density coincided with fine organic ooze while minimum density was found in sand.

In the Chesapeake Bay, milfoil has generally been considered to grow best in soft muck or sandy muck bottoms (Anderson 1972; Steenis et al. 1967; Philipp and Brown 1965; Springer 1959). However, milfoil is capable of invading hard sand in protected areas (Steenis and Stotts 1961) and in oyster beds (Rawls 1964, 1965a, 1975).

Light, Depth and Turbidity. The limiting or lethal extremes of depth, turbidity and light as they apply to milfoil are not clear. The penetration of light into water is directly proportional to both depth and turbidity, and turbidity in turn is a reflection not only of sedimentation rates, but wave, tidal and wind action, erosion, and a multiplicity of environmental factors encountered under field conditions.

From a production study in Lake Wingra, Wisconsin, Titus et al. (1975) concluded that light and temperature were primary environmental forcing functions. Plant growth was limited in extremely turbid waters to 1.5 m, but plants reaching the surface were able to slough off the lower leaves and thus better adapt to turbid waters. Patten (1956) felt that in Lake Musconetcong, New Jersey, where milfoil was found in waters to 2 m deep, water depth and temperature were secondary in determining milfoil density. In studying the relation of depth to temperature, Grace and Tilly (1976) looked at three stations in a reactor cooling reservoir. Rooting depth was greater at the cold station than the hot: 4 to 5 m at the cold; 4 m at the warm; and 3 m at the hot station. Light penetration may have been partially responsible.

In the Chesapeake Bay milfoil has been found to require moderately high light intensity, is sensitive to turbidity and grows in water more than 2 m deep, if clear (Southwick 1972). Springer et al. (1961) reported milfoil in clear water to 3.5 m deep at high tide, and in 1.8 to 2.1 m when the Secchi disc reading was 0.8 m. Rawls (1964) found milfoil growing in clear waters from a few centimeters deep to 2.7 m deep at mean low tide (tidal amplitude in the upper Chesapeake Bay areas studied averaged about 0.5 m). Southwick (1967-1969) felt that low light and turbidity might have been the main limiting factors for milfoil establishment in Back River at Baltimore. Rawls (1971b) during field studies in Chaptico Bay, found dense milfoil beds in waters where Secchi disc readings ranged from 0.5 to 1.2 m. During the period of this study, carp-rooting action kept the water extremely turbid in fingers and small guts, but milfoil was present in moderate to heavy stands and extended to the surface at high tide. A Secchi reading in these areas at such times of carp activity was about 7.5 cm.

Kerwin et al. (1975b) assumed turbidity along with salinity to be major limiting factors affecting the distribution and abundance of vascular aquatic plants in the Chesapeake Bay. Rawls et al. (1975) pointed out such activities as bank and upland erosion, construction, dredging, increased boating, stream channelization and straightening and eutrophication were contributing factors to turbidity increases.

Current, Wind and Wave Action. Milfoil grows best in protected waters but can withstand tidal fluctuations of 0.6 m (Springer 1959). Milfoil is vulnerable to strong tidal currents and wave action that might not affect other native, rooted submersed aquatic vegetation species (Rawls et al. 1975). Severe wave action limits milfoil establishment, but once established, milfoil helps reduce wave effects (Stotts 1961).

Sedimentation. Grace and Tilly (1976) list Sculthorpe (1967) in their literature cited, and attribute him with having generalized that milfoil is likely to be eliminated from regions where suspended solids tend to settle at high rates.

Nutrient Response. The growth of aquatic weeds, including Eurasian watermilfoil, is promoted through enrichment of habitat waters by nutrient materials, mainly nitrogen and phosphorous. The chief sources of these nutrients are fertilizers, sewage effluents, disposal plants, overflow and seepage from septic tanks, waste from pleasure craft and other vessels and migrating ducks and geese (Elser 1966).

Dr. Clyde Reed pointed out (Beaven 1962) that in Maryland, all waters that were heavily infested by milfoil received their runoff from limestone areas. This was also true in T.V.A. lakes and other areas where milfoil was abundant. With a reduction in coal mining in Pennsylvania, acid wastes finding their way into the Susquehanna River at the head of the Chesapeake Bay decreased and calcium ions sharply increased in recent years. This has favored growth of milfoil in the upper Bay. John Gallagher reported (Beaven 1962) that the milfoil problem was severe in northern New Jersey where the presence of iron was high. Anderson (1964) found no significant increase in minerals in milfoil, despite mineral increases in waters from which the milfoil was taken. He felt that milfoil could use the bicarbonate ion directly from the water for its carbon source since carbon levels were almost identical in milfoil taken from fresh waters as milfoil taken from brackish waters. Potassium and sodium were concentrated in plants taken from fresh water, but not in plants from brackish waters. The magnesium ion increased in the plant as it increased in the water.

In a New Jersey study using plastic ponds and pond water, Ryan et al. (1972) investigated the effects of fertilization on milfoil, Elodea canadensis and Potamogeton pulcher. The addition of NH_4, NO_3, phosphorus and potassium added nothing to plant growth the first year. In the second year, P. pulcher benefited, but milfoil and Elodea grew better in a control environment, possibly due to no competition from algae. Mulligan et al. (1976) experimented with nitrogen and phosphorus fertilization on Elodea, milfoil (Myriophyllum spicatum var. exalbescens), Ceratophyllum demersum, Potamogeton crispus and algae in 0.004 ha experimental ponds at Cornell, New York. High fertilization tended to eliminate benthic plant populations and decrease the standing crop of phytoplankton. Elodea grew in the high nutrient levels, but Ceratophyllum and Myriophyllum were eliminated. Amonium nitrate and triple super phosphate were used in weekly applications. Low fertilization rates were 0.05 mg nitrogen/l and 0.005 phosphorus/l; high rates were 100 times the low amounts.

Susceptibility and Control. Due to extensive research initiated to determine effective methods of control for Myriophyllum spicatum, a strong data base is available concerning elements,compounds and organisms that negatively impact milfoil. Natural physical controls such as sedimentation, light, turbidity and salinity have been discussed previously. Some are quite effective. For instance, in North Carolina at Pea Island Refuge, just south of Currituck Sound, milfoil beds were decimated by salt water intrusion in 1962 and by 1967 had not regained problem status (Crowell et al. 1967).

Bio-controls may cause reduction of milfoil beds also. In 1963, when milfoil began to disappear in many areas, some milfoil beds in the Potomac River were so smothered by blankets of diatomaceous algal growth, that milfoil photosynthesis would have to have been highly reduced, if even possible. Other plant species associated with milfoil were similarly affected, (e.g., Potamogeton perfoliatus) (Rawls 1964; Steenis and King 1964). A minor controlling influence might have been barnacles which set on milfoil stems, virtually weighting them to the bottom, or furnishing resistance to mechanical actions of tides and winds until the stems became broken. However, since only a node is required to begin a new plant, thinning of milfoil in such a fashion would be strictly localized and distribution possibly even enhanced.

Boyer (1960), while experimenting upon milfoil in tubs, found that after 25 days such an intense diatom bloom developed that milfoil growth was inhibited. Boylen and Brock (1974) identified the diatom Cymbella as epiphytic on stems and leaves of milfoil decaying under ice in Lake Wingra, Wisconsin. They also found a heavy epiphytic mat of Oedogonium sp. on milfoil from June to September. Sparrow (1974) reported a species of the fungus Physoderma found on M. spicatum in Squaw Lake, Michigan in 1972.

The Agricultural Research Service through Public Law 480 has sponsored research on bio-control by insects on plant hosts. Larvae of the moth, Parponyx stratiotata and the weevil (Litodactylus leucogaster) prefer to feed on milfoil. Another moth species, *P. allionalis*, in Florida seems to have similar habits, but further study is needed (Sailer 1972).

Pumpkinseed (*Lepomis gibbosus*) was seen (Rawls, personal observation) in shallow shoreline waters uprooting milfoil by nosing and finning it out of the substrate until an area large enough to accommodate their nest had been cleared. When many nests occupied a small area, the small depressions were noticeable in an otherwise unbroken band of milfoil. Carp, noted for rooting activities and creation of turbid conditions, can clear an area of milfoil. But by fragmentation, both carp and pumpkinseed can indirectly aid in milfoil distribution Steenis and King 1964). Rawls et al. (1975) gave examples of carp destructiveness. Almost 5,000 ha of aquatics were eliminated by carp in the late 1950s on on the Susquehanna Flats. In Marshall Creek, off Nomini Creek, a Virginia tributary to the lower Potomac River, a surface mat of milfoil was eradicated by carp in the last two weeks of May, 1968.

Much research has been conducted on a possible disease control of milfoil. One of the earliest observations that subsequently led to studies of the "Lake Venice" disease of Eurasian watermilfoil was made by the late Harold Elser (Haven and Wass 1963). Elser watched the reduction of milfoil in Lake Venice, Maryland, during the summer of 1962, and suggested a natural control might be operating. Elser (1966) said that there might actually be two diseases involved but since the phenomena affecting the plants were not definitely known to be diseases, perhaps "valetudinous plants" would be a better word than disease. The Lake Venice disease had reduced the plant cover from 100 percent to 20 percent by 1963, and to 10 percent by 1964.

The chief characteristic of the disease was a heavy overgrowth of diatoms, epiphytic algae and various sessile protozoans. These organisms became so thick that along with the silt they collected, the leaflets became entirely obscured. Elser did not know whether the overgrowth was the disease or the symptom. By 1964, the so-called Lake Venice disease was seen almost everywhere in the Bay, usually affecting large areas.

The second disease was first noticed by Maryland biologists Vernon Stotts and Guy Lerner in the Northeast River, tributary to the northeast corner of the Susquehanna Flats at the head of the Chesapeake Bay. The condition was characterized by a stiffening of the stem and leaves. As the disease advanced, the leaves fell off, leaving stiff bare stems which stood out of the water at low tide. However, the roots apparently were not affected because new growth started even before all the dead stems were gone. In late June of 1964, the extensive beds of milfoil in the upper western shore Bay tributaries (Seneca, Salt Peter and Dundee Creeks) had all but disappeared, but by the first of August, new growth again had reached the surface. By 1964, the two diseases were so widespread that it was difficult to find four tons of healthy milfoil for experiments.

The Department of Chesapeake Bay Affairs, as reported by Johnson (1966), estimated that by 1966 the total milfoil remaining in the Bay was less than 15 percent of the 40,500 ha estimated to be present in 1963, presumably because of the diseases. Elser was quoted as having said, "However, during the last week of July many beds of milfoil suddenly showed a very rapid growth of new stems.

This new growth flowered profusely, a phenomenon which had not been observed since 1964. The new, lush growth of milfoil remained healthy for about two weeks when definite symptoms of the disease again appeared. By the middle of August, the plants again were seriously infected and it appears that beds will be further diminished."

Elser (1967) stated that by 1965, only 5 percent of the original milfoil acreage remained in Lake Venice. There was a definite lack of flowering on diseased specimens, but in July, 1966, with the explosive growth from old stems (up to 10 cm per day), flowering began immediately.

Bayley et al. (1968) believed pathologic conditions were responsible for the decline of milfoil. Northeast disease symptoms were transmitted in the laboratory by a bacteriologically sterile filtrate passed through a 0.2 micron filter. This could indicate a virus, virus-like particle or a toxin as the etiologic agent. Gram-negative bacilli obtained from diseased milfoil probably represented a secondary infection. Because diseased milfoil in some Bay areas showed a resurgency of growth in 1967, and successfully flowered, it was thought possibly to indicate development of genetic resistance to the pathogen, or a natural cyclic pattern with reduced pathogen virulence.

Southwick (1967-1969) detailed in quarterly progress reports studies of the Northeast disease of milfoil. In 1968, infectivity of an active filtrate from diseased milfoil was significantly reduced by heating to 70 C for 30 minutes. This was considered as evidence of virus. Electron microscopy of a filtrate from diseased plants revealed virus particles 80 millimicrons in length which resembled the potato yellow dwarf virus in morphology, though smaller. In 1969, Southwick (1967-1969) was unable to detect the milfoil virus by physical tests on frozen preparations of diseased milfoil. This indicated an agent unstable to freezing and storage and not easily characterized by standard virological techniques. In a later 1969 progress report, Southwick was unable to isolate the virus and demonstrate infectivity as in 1967 and 1968. He believed the main wave of the virus had passed but the symptoms persisted. He was unable to infect milfoil with other common plant viruses. In the last progress report of 1969, Southwick wrote that milfoil had disappeared from the three study areas, presumably the Back, Middle and Rhode Rivers. The complete loss was attributed to diseased conditions in September. He reported that milfoil had continued to decline the last four years in the Middle and Rhode Rivers and that summer growth was not persistent.

Steenis (1970) reported that no virus-type disease factors had been noted in the Currituck Sound area, that attempts to introduce the disease into TVA lakes were unsuccessful and that symptoms had been noted in the Chassahowitska area in Florida, but not in the adjoining Homossassa and Crystal Bay localities. Attempts by Bayley and Weldon to introduce the disease into Crystal River also failed. Steenis (1970) pointed out that the disease factors accentuated vulnerability to adverse environmental conditions.

Bean et al. (1973) performed a laboratory study on Lake Venice disease. They were not able to produce diseased plants by direct innoculation of healthy plants or by growing diseased and healthy plants together. However, plants under low light intensity innoculated with extracts of diseased tissues produced symptoms resembling Lake Venice disease. They believed this disease to occur under stress conditions and to increase plant suceptibility to attack by microorganisms.

A natural or biological control of a pest plant is preferred to the introduction of herbicides into waters which frequently are already over-loaded with undesirable pollutants. Usually bio-controls are not known or are not immediately available, so often the urgent need to contain or restrict noxious vegetation requires the careful and immediate use of an herbicide. Also required is knowledge of what effects such an herbicide has on non-target organisms and the general environmental quality surrounding the target species.

Steenis (1966) referred to some of the difficulties in determining the effectiveness of a herbicide. Laboratory screening followed by field testing is requisite, but one is not necessarily translatible to the other. And a herbicide that is effective on one plant species at a given dosage rate at a certain time and in a particular geographical location with defined chemical, physical and biological characteristics may not be effective a short distance away where an application is made within minutes of the previous application and where environmental parameters appear to be virtually identical. In New Jersey, milfoil's vulnerable period to 2,4-D is from mid-May into July, while in the Chesapeake Bay, the vulnerable period to similar amounts of the same herbicide extends from the last 10 days of May into the first week of June. Apparently, anthesis and density of the milfoil blanket account for the phenological difference.

Sodium arsenite at one time was widely used to control aquatic vegetation. In an Iowa lake, Myriophyllum spicatum and other aquatics were successfully treated from the air and by boat with this chemical at concentrations of 3 to 5 ppm. Fishes were not adversely affected, but some clams (Anodonta marginata) and cray fish were killed (Rose 1955).

Through the years other chemicals have been studied. Peverly and Crawford (1975) used glyphosphate at up to 5,000 ppm on milfoil, but the solution was ineffective in preventing root regrowth. In 1958, Steenis began studies with 2,4-D, Fenac and silvex in weighted emulsions and granular formations, and learned that control was hampered and complicated by tidal water movements (Springer and Stewart 1959). Springer et al.(1961) reported that the granular formulations of 2,4-D esters at 22 to 44 kg/ha a.e. gave good results and that 2,4-D was less expensive and more specific than silvex, liquid or granular. Steenis and Stotts (1961) found that of 16 herbicides tried in various concentrations, 2,4-D as noted above was most effective in tidal areas and should be applied when the tide was ebbing or near low water slack, after milfoil reached the surface and the water temperature was above 20 C. Steenis and Stotts (1965) stated that diquat was picked up more rapidly than 2,4-D and that better dispersal was obtained with it, but that its effectiveness was influenced by various factors such as roiled water and algal-diatomaceous ooze on the plants.

Beaven (1962) wrote that the most consistent control of milfoil was obtained with esters of 2,4-D (butoxyethanol, iso-octyl, and propylene gylcol to butyl ether). Dosage rates at 22 kg/ha a.e. yielded almost complete control, and when water movement was limited, the amine salt of 2,4-D at 11 kg/ha a.e. gave similar results. Herbicides were impregnated on 8 to 15 mesh baked attaclay granules and dispersed by an air blower through a hose mounted on the bow of an outboard motor boat.

Stanley (1974) found in a laboratory study that the effectiveness of 2,4-D was twice as great on milfoil growing in soil than on milfoil growing in sand. Calcium chloride added to sand cultures caused a comparable or greater increase in 2,4-D effectiveness than sand cultures without calcium chloride. In TVA field treatments there was up to 48 percent difference in effectiveness between areas that had the highest natural calcium content and the lowest.

In laboratory experiments, Stanley (1974) investigated the synergistic effects of combining 2,4-D with other compounds. He found that control effectiveness increased with $HgCl_2$, $AlCl_3$, NaCl, $NaAsO_2$ and $CuSO_4$. NaCl as a synergist, however, required concentrations so high that it was economically and environmentally unfeasible. Stanley felt that this high concentration demand might account for the erratic results in field tests of 2,4-D on milfoil in estuaries subject to salinity fluctuations.

Controlling milfoil by shutting off light from the plant has been studied. Smith (1962), in TVA lakes, experimented with black polyethylene covers held in place over milfoil beds with anchors and floats. In 72 days, this complete shading killed the plants, while a 21 day period did not (Beaven 1962; Smith 1962). Elser (1967) stated that a black aniline dye had been successfully used in Arizona in 1947 to shade out plants. It was quite successful, but required a year to produce results. In 1965 to 1966, experimental use was made of black plastic anchored and weighted to the bottom or floated over the top of milfoil beds in the South and Sassafras Rivers. From these experiments, Elser (1967) concluded that black plastic floated on the surface for a two-week period early in June would give weed control for the entire summer.

Mechanical control (underwater cutting, mowing, scraping, dragging, water-jetting with numerous variations and combinations) has also been used in milfoil erradication attempts. Springer (1959) mentioned that cutting about 15 cm above the bottom with a small, horizontally-mounted buzz saw made possible the temporary clearance of about 0.5 ha day at a daily cost of $40 (1959 figures). Revegetation of the cut area occurred during the same or following season but the cut portions created new infestations. Elser (1966, 1967) described different devices used for cutting and/or removing aquatic weeds: hydraulic jets; a 15 m steel ribbon with saw teeth on both sides and wooden handles on the ends; and a V-shaped blade dragged along the bottom. He also mentioned that the Department of Chesapeake Bay Affairs owned a large harvester which, though it did a good job, would have taken 37 years to get around once to all the Bay's weed beds, even if the machine would work all year-round. Mowing, at best, Elser concluded, was a temporary measure. Rawls (1964, 1975) concluded that mowing alone was effective only for a season and had to be repeated monthly, depending on the type of activity necessitating the mowing. Cutting followed by 2,4-D application at usual rates extended milfoil's vulnerable period beyond anthesis, but the control effects were scarcely noticeable the following season. Mowing seemed to be practical only for small areas such as private boat docks, small marinas, opening short channels to open water, etc.

As previously pointed out, salt water intrusion decimated Eurasian watermilfoil in the lower Potomac River in 1933 and at Pea Island National Wildlife Refuge in 1962. Knowles (1976) developed two models evaluating control through salinity manipulation in Currituck Sound. Economically, neither the pumping in of salt water or the opening of an inlet to the sea seemed feasible. Also, if salinity were to be raised, other fresh water macrophytes would probably be excluded.

A further control method that has met with success in TVA lakes is water level manipulation along with herbicide application. However, conditions within the Chesapeake Bay estuary preclude the use of such measures.

Productivity

Productivity of Myriophyllum in the natural environment is influenced by a host of environmental factors. Inclusive are carbon sources and concentrations, light, temperature and salinity. Variations in these factors may affect productivity alone or variations in two or more factors may produce synergistic or antagonistic interactions.

Myriophyllum is able to utilize the bicarbonate ion as a carbon source (Anderson 1964; Hutchinson 1970, cited in Nichols 1975; Van et al. 1976) though free CO_2 is the preferred form (Van et al. 1976; Steemann Nielson 1951). Milfoil can also utilize half-bound CO_2 (Patten 1956). Carbon sources are absorbed by both the upper and lower leaf surfaces. The absorption and subsequent translocation is linearly dependent on the ion concentration up to a concentration of 10^{-3}M $CO_2(HCO_3^-)$ /l under optimum light conditions. There is a decrease in HCO_3^- adsorption when Myriophyllum is switched from a balanced salt solution to a pure solution of $KHCO_3$, $NaHCO_3$ or $Ca(HCO_3)_2$ with the same HCO_3^- concentration. Photosynthesis rates respond rapidly to changes in aqueous concentrations of carbon with decreasing photosynthesis as carbon levels decrease (McCraken et al. 1975).

Hydrogen ion concentration indirectly affects photosynthesis by affecting the carbon concentration. Milfoil photosynthesis, when taken from pH 3.1 to 9.3, is unduly affected by the reduction in free CO_2 (Van et al. 1976); however, Shiyan and Merezhko (1972) found the efficiency of milfoil to be 10 times higher in alkaline solutions than in acid conditions provided in two had equal CO_2 concentrations.

Titus et al. (1975) pointed out that milfoil has the potential of becoming a plant nuisance in thermally enriched environments, probably due to high optimum temperatures required for milfoil photosynthesis.

Several authors have reported on the effects of light on photosynthesis. Steemann Nielson (1951) in a laboratory study with milfoil grown in optimum light and in optimum HCO_3^- concentration showed that photosynthesis was not limited by a photochemical reaction. In Lake Wingra, Wisconsin, Adams et al. (1974) found that at a depth of 2.4 m, 56 percent of the photosynthetic activity occurred within 1.0 m of the surface in May. In August, 57 percent occurred within only 20 cm of the surface. They concluded that light and depth distribution of photosynthetic tissue was most important in causing variation in photosynthesis. McCracken et al. (1975) stated that photosynthesis peaked at mid-day or shortly thereafter.

Photosynthesis, but not respiration, appears to be affected by salinity although there seems to be a controversy. Without giving quantitative salinity measurements, Boyer (1960) stated that there was no difference in milfoil respiration and photosynthesis in fresh or saline waters. However, in the laboratory, McGahee and Davis (1971) demonstrated a depression in photosynthesis when apical portions of Myriophyllum were taken from lower to higher salinities and whole plants were killed or stunted at 13 to 14 ppt. Graham and Davis (1972) found that photosynthesis was inhibited at 32 ppt after 20 hours but that respiration was unaffected. After 10 days at 16 ppt photosynthesis was low. At salinities of 4, 8, 16 and 32 ppt, net photosynthesis was found to decrease with increased salinity in light bottles, but again respiration was little affected in dark bottles. The effect was found to be reversed if $CaCl_2$ was added to high concentrations (1:22 and 0.1:22 Ca:Na). At lower concentrations (0:22 and 0.01:22 Ca:Na), milfoil cells were observed to disintegrate and the membranes ruptured (Davis et al. 1974).

McGahee and Davis (1971) studied interactions between light and depth in regard to photosynthesis of apical portions of Myriophyllum collected in water 0 ppt salinity. Under 20 hours of low light in constant illumination and in 32 ppt, photosynthesis was depressed but neither photosynthesis nor respiration was affected when salinity was lowered. With moderate light in a light/dark regime of 10 hours/14 hours, photosynthesis was inhibited at 16 ppt and respiration remained high.

Milfoil grows rapidly. In the Chesapeake Bay, during its mid-summer production peak, it can grow 2.5 cm a day after being severed by an underwater mower (Rawls 1964), or after being cut in other growth periods, as much as 45 cm per month from a new node (Rawls 1975). Smith (1962) found 1 m of growth in one season from a 7 cm fragment in a TVA lake. In turbid waters where the lower leaves are sloughed off, the same biomass tends to become concentrated in the upper canopy. Loss of lower leaves also occurs in non-turbid waters of the Chesapeake, and appears to be characteristic of Myriophllum spicatum particularly in dense beds (Rawls, personal observation).

Adams et al. (1974) determined that the maximum ash-free dry weight of a standing crop of milfoil (220 g/m^2) was about 100 g C/m^2. Nicholson and Post (1975) stated that ash content was required to estimate organic production and possible economic uses. Ash content increased with alkalinity in the Haloragaceae but not in some other aquatic plant families.

Consumer Utilization

As with most submersed aquatic vegetation species, Eurasian watermilfoil provides some or all of the basic life requirements--food, shelter, protection, nesting and resting--for animals and to a lesser extent, plant species associated with it or coming in contact with it.

Many of the minnows and larger fish are known to deposit eggs on submersed vegetation. When plant species serving this purpose are listed, milfoil is frequently among them. Rawls (1975) observed fishermen casting for largemouth black bass (*Micropterus salmoides*) along the edges of experimental plots cleared of milfoil in Dundee Creek. The fishermen reported excellent catches as long as plots remained open. Crowell et al. (1967) reported that in the Currituck Sound-Back Bay area, an increase in sport fishermen catch rates occurred until milfoil beds became so dense the edge effect was lost.

Patten (1956) stated that waterfowl feed upon milfoil achenes, and from his literature cited, mentioned that Guppy in 1897 suggested that seed germination might be increased by passage through the alimentary system of birds. Patten (1956) concluded that waterfowl were probably the most important dispersal agents of fruits to various drainage systems. However in food habit studies in the Chesapeake Bay (Rawls, in press), most of the milfoil seeds in waterfowl gizzards were crushed and broken by the powerful grinding action of the gizzard, unless the seeds were freshly ingested. Patten (1956) suggested that the small, slime-covered spring buds or turions were able to adhere to waterfowl feet because of a glucosidic envelope and provided a potential source for milfoil distribution into new areas.

Springer and Stewart (1959) reported that Coot (*Fulica americana*), Wigeon (*Anas americana*) and probably Gadwall (*Anas strepera*) have been observed feeding on milfoil. Crowell et al. (1967) discovered that apparent feeding by waterfowl on milfoil in the Currituck Sound-Back Bay area was deceiving. Milfoil was growing in the same type habitat as were preferred plant foods and after about the third year, as desirable foods were crowded out by milfoil, waterfowl left and moved to areas where choice foods still existed.

Bayley et al. (in press) observed that waterfowl numbers decreased on the Susquehanna Flats during the period of greatest milfoil abundance. Martin et al. (1951) classified milfoil as a low grade duck food. However, though the quantity of woody seeds eaten was usually small, the nutlets were eaten to a considerable extent in the prairie lakes of the Dakotas and in adjoining states. Florschutz (1973) examined the contents of 170 waterfowl digestive tracts collected in the vicinity of Back Bay and Currituck Sound during hunting seasons from 1968 to 1971. Included were 27 Canada Geese, 6 species of dabbling ducks, and 4 species of diving ducks, plus 31 Coots. Milfoil was found in 71.8 percent of the birds and comprised about 33 percent of the total food volume. The highest milfoil use was noted in Scaups. Rawls (in press) examined 2,747 waterfowl gizzards taken during hunting seasons in the Chesapeake Bay area during 1959 to 1968. Represented were 2 species of geese, 6 species of dabblers, 8 species of divers and 1 species of Merganser. Slightly over 10 percent contained milfoil seed, stems or leaves. In volume, milfoil averaged 4.2 percent by volume of all foods eaten, and per individual bird, ranged from zero or trace amounts to 100 percent. Wigeons used milfoil to the greatest extent (over 21 percent) of all birds examined.

Milfoil offers attachment sites for organisms which later become food for higher life forms (Springer 1959; Springer et al. 1961). Patten (1956) found that milfoil offered support for attached Aufwuchs and detached organisms such as tychoplankton and that milfoil, because of its finely dissected leaves, could support more periphyton than most other aquatic phanerogams.

Economic Uses and Problems

The redeeming qualities of milfoil are not as noticeable as the problems it causes. However, Myriophyllum spicatum is not totally devoid of positive features. Springer (1959) noted that milfoil could occupy sites which otherwise would be inhabited by objectionable algae. Nichols (1975) found that milfoil was a good hydrosol stabilizer, and as animal food, had a high xanthophyll content and was comparable to alfalfa in protein content. However, Haven and Wass (1963) noted that milfoil xanthophyll was unstable. Haven and Wass (1963) reported that the amounts of Ca, K, Na, N, P, and Mg were insufficient to class milfoil as a good source of nutritive elements.

Elser (1966) reported that the high ash content of milfoil precluded its use as a chicken food because it gave the birds diarrhea. As cattle food, it was about as good as high quality hay (about 19 percent protein). Milfoil's chief drawback was its water content of 90 percent . Elser (1966) added that one Annapolis high school girl made cookies of milfoil flour and pronounced them good. Elser (1967) wrote that milfoil's N-P-K value was 3-2-5. This was better than cow (2-1-1) or sheep (1-1-2) manure (Elser 1965).

However, the story of milfoil is concerned less with pleasure and more with vexation ranging from mildly annoying to verging on economic disaster. In order to cope with ramifications of the milfoil explosion, an interagency cooperative effort of personnel representing Federal, state, private, educational, commercial and industrial interests plus representatives of the general public met annually from 1959 through 1964 to discuss any and all aspects of milfoil infestation not only in Maryland, but in other affected areas. Agencies and their representatives most active in milfoil investigations in the Chesapeake Bay area were: U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center (John Steenis); Maryland Department of Natural Resources Wildlife Administration (Vernon Stotts); Virginia Institute of Marine Science (Dexter Haven); University of Maryland, Chesapeake Biological Laboratory (Charles K. Rawls, Gordon Beckett, Theordore Will and G. Francis Beaven).

By early 1963 (Kelly 1963), milfoil had become such a problem that a noxious weed bill H.R. 2994 (Morton) was introducted in the House of Representatives. The bill provided for the controland progressive eradication of certain aquatic plants in Maryland, Virginia, New Jersey and Tennessee. It was concerned with Eurasian watermilfoil and involved Federal and state cooperation through the U.S. Army Corps of Engineers.

The impacts of the milfoil explosion were felt in many areas of the Chesapeake Bay. Waterfowl distribution was affected because winter feeding grounds, such as the Susquehanna Flats, seemed to lose their attractiveness to many species of water birds, among them, Canvasback (Aythya vallisneria), a threatened species. Springer et al. (1961) identified some of the aquatic plants favored by waterfowl for food which milfoil chokes out: Potamogeton pectinatus Vallisneria americana and Ruppia maritima. Steenis and Stotts (1961) noted that oxygen was reduced under the heavy, surface milfoil blanket.

Milfoil interferes with oyster (Crassostrea virginica) growth by reducing water movement and restricting flow of food to oysters. Lack of water movement also permits surface water temperatures to increase to the point they may have detrimental effects to plants and animals in shallow waters. Anaerobic conditions arise as a result of decomposing milfoil, and sulphur bacteria then release hydrogen sulfide, killing the oysters. In addition, dense mats make harvest almost impossible by dredge and tongs.

Milfoil seriously interferes with angling, with fishing by means of haul seines or fyke nets and with crabbing by pots, trot lines and scrapers. Milfoil also restricts navigation for commercial fisheries, curtails pleasure boating, water skiing, swimming and clogs water intakes in pumps and motors. It contributes to a more rapid build-up of the bottom through deposition of organic material and facilitation of silting. Milfoil beds appear to provide conditions suitable for mosquito (Anopheles quadrimaculatus) and possibly certain Culex sp. Undesirable odors are created by windrows of decomposing vegetation deposited by wind and tide on shorelines. The possibility exists that milfoil can impart off-tastes to the water and it may lower nearby real estate values.

ZOSTERA MARINA[2]

Biology

General Morphology. Zostera marina is one of the few species of aquatic plants to inhabit the marine environment (Hartog 1970). It is a true flowering plant belonging to the phylum Spermatophyta and to the class Monocotyledoneae or plants with embryos with a single cotyledon. It is a member of the pondweed family, the Potamogetonaceae. The plant is known commonly by many names, the most widely used being eelgrass, but sometimes referred to as grass wrack. The name Zostera comes from the Greek word Zoster, a belt, in reference to the long, blade-like leaves of the sea.

Eelgrass has a creeping rhizome, 2 to 5 mm thick, with numerous roots and a leaf at each node. Internodes are 10 to 35 mm long. The tubular, membranous leaf sheath may be 5 to 20 cm long and is wider than the leaf-blade. The leaf-blade may be up to 120 cm long and 2 to 12 mm wide with 5 to 11 nerves (Hartog 1970) (see Figure 7).

The leaf-blade of eelgrass displays a variety of morphological forms depending on the depth of the water, salinity, temperature, current regimes and sedimentological characteristics (sand, mud, related nutrients) (Orth, in press; Ostenfeld 1905; Butcher 1935; Burkholder and Doheny 1968; Harrison and Mann 1975 Hartog 1970; Setchell 1927, 1929; Phillips and Grant 1965; Phillips 1972; McRoy 1966, 1970a, 1970b; Philip 1936). In general, eelgrass has usually displayed the small narrow-leaf form in shallow, more physically exposed, sandy substrates

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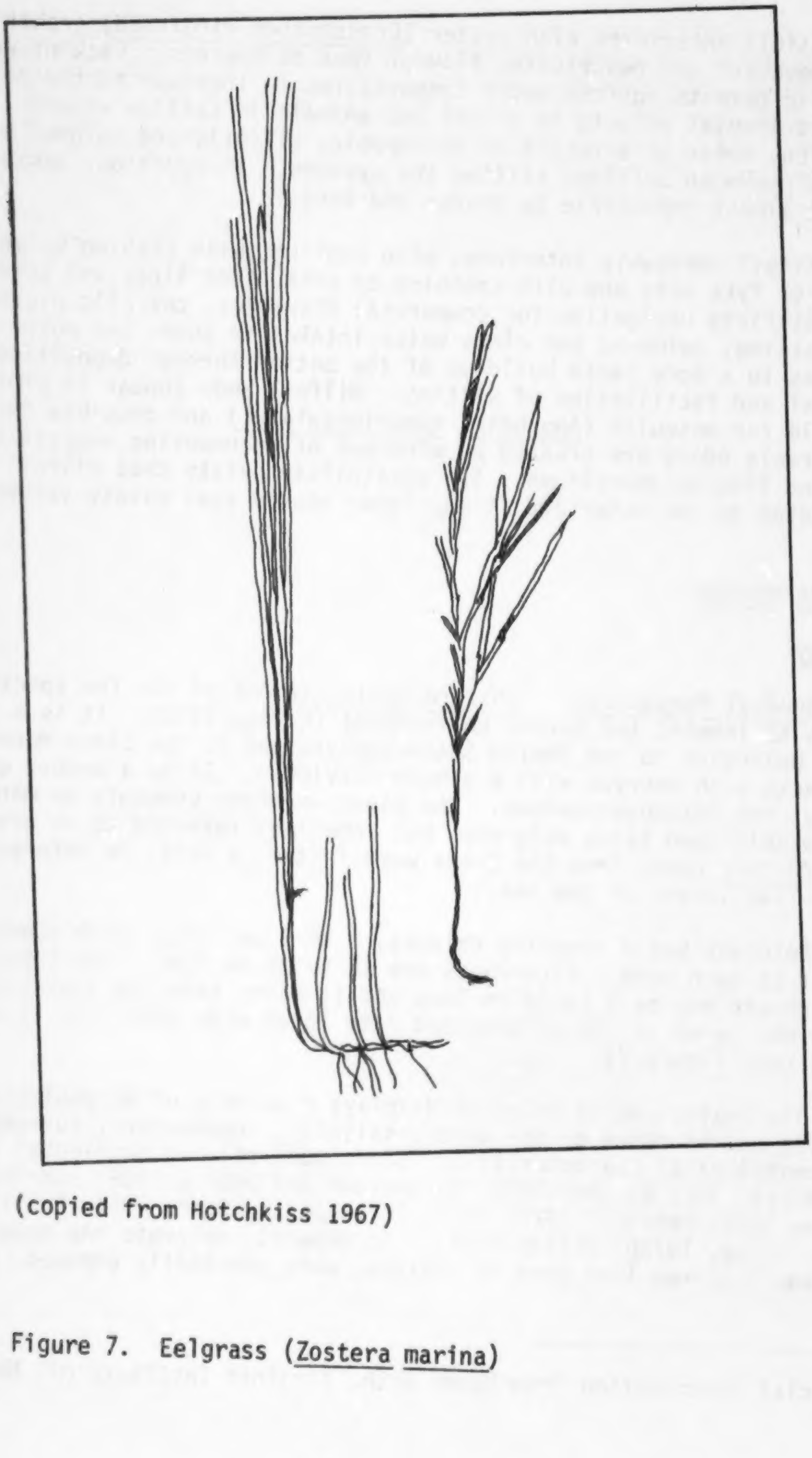
[2] Special contribution from Rober Orth, Virginia Institute of Marine Science



(copied from Hotchkiss 1967)

Figure 7. Eelgrass (Zostera marina)

while in deeper muddy and/or less exposed areas, the longer, wider form predominates. These characteristics are typical of the eelgrass found in various locations in the Chesapeake Bay with one exception. The eelgrass at Tangier Island is exceedingly lush (turions over 1 m long in water only 0.3 m at MLW) and may be a response to nutrient addition from the adjacent, low lying, unsewered community. Orth (in press) found that the addition of commercial fertilizer increased the growth of eelgrass significantly (both length of turions and biomass) in a bed on the Eastern Shore of Virginia and suggested that the growth form may be a function of the amount of nutrients in the sediment. Sedimentary nutrient content may in turn be directly related to sediment type (i.e. mud or sand).

This difference in growth form has created some taxonomic problems in that two species of eelgrass have been described, one for each form (Hartog 1970). However, Phillips (1972, 1974b) has demonstrated via transplants, that eelgrass maintains phenotypic plasticity and conforms to the conditions of its environment. One interesting observation is that of Blackburn (1934) who noted that after the wasting disease in the 1930s in England the narrow-leaf form returned in areas where the wide leaf form had previously been found. The drastic change in sediment type after the disappearance of eelgrass (mud to sand and associated nutrient changes) may have resulted in the change in form of the grass. Similar observations have been made in the York River, Virginia, after the disappearance of eelgrass in this area (Orth, personal observation). The eelgrass inhabiting these areas, in what were once extensive meadows, is of the short, narrow leafed variety where, prior to the loss, the long, wide-leaf form was prevalent. The sediments in this area have been drastically altered after the removal of eelgrass (Orth 1975a).

Reproduction. Vegetative propagation as in most other aquatic plants occurs via elongation and growth of the rhizome. Sexual reproduction occurs with the formation of generative shoots which are erect branches born from the prostrate perennial rhizome. These shoots are normally terminal on the rhizome (Churchill, in press; McRoy 1966; Orth, personal observation; Setchell 1929) but may appear on lateral branches (Phillips 1972). Reproductive turions may represent 3 to 22 percent of the total number of turions (Churchill, is press; McRoy 1966, 1970a; Phillips 1972; Sand-Jensen 1975; Orth, unpublished) but may be as much as 100 percent of the turions (Felger and McRoy 1975).

Eelgrass is monoecious, bearing both stamens and pistils in structures called spathes, born on the reproductive shoots. There may be from 1 to 25 spathes per turion (see all references in first paragraph of this section). Pollination occurs with the aid of water currents. The stigmas of the pistils protrude through the spadix opening and are pollinated by contact with the drifting, threadlike pollen grains (Phillips 1972). Flowering and fertilization occur in the spring or summer (depending on geographical location — spring in the Chesapeake Bay) and self-fertilization is prevented by protandrous development of stamens and pistils. The fruits of eelgrass do not have any special adaptation for dispersal. Usually, the generative shoots become detached at the time of fruiting and float to the surface where they are carried by currents. Ripe fruits are shed as the shoots are carried along the surface. However, it may be that many of the seeds are released in the bed before the generative shoots break off.

Sexual reproduction is apparently controlled by temperature (Setchell 1929; McRoy 1966) though day length may be highly integrated into this activity (Phillips 1974a; Harrison and Mann 1975). Setchell (1929) proposed that flowering occurred between 15 to 20 C although some authors have found flowers in waters less than this temperature (Phillips 1972, 1974a ; Harrison and Mann 1975). Setchell (1929) and McRoy (1966) have noted that in one particular area, eelgrass in shallow water, which warmed more rapidly and reached a slightly higher temperature than a deeper area, flowered while that in a deeper area, where temperatures remained cooler, never flowered. In the Chesapeake Bay, eelgrass has been observed flowering in shallow water (<0.1 m MLW) almost one month before reproductive structures were found in a adjacent deeper area (1 m MLW) (Orth, personal observation).

Germination of seeds is also related to temperature. In the Chesapeake Bay, seeds (sampled from monthly sediment samples taken in an eelgrass bed) apparenly germinate in the fall as temperature declines (Orth 1975b, 1976). Reports from more northern locations indicate some seed germination in the fall with the majority of the germination occurring the following spring (Tutin 1938; Addy 1947a, 1947b; Arasaki 1950b; Taylor 1957b; Churchill, personal communication). However, these studies failed to look at seeds from sediment samples in the beds and based their observations on visual sightings of new leaves emerging from the substrate. Thus, germination may actually occur in the fall. However, if temperature is a critical factor (e.g. 10 C) for germination and seeds are not released from the plant until temperatures are at or below this critical temperature, then germination may occur in the spring, as temperature begins to rise to that critical temperature. Phillips (1972) found that seed germination occurred throughout the year but with the major occurrence between April and July. Setchell (1929), Tutin (1938) and Taylor (1957a,b) gave a complete description of embryo and seed development.

Distribution

Historically, eelgrass has been the dominant submerged aquatic vegetation in the mesohaline and polyhaline regions of the Bay and its subestuaries. During the 1970s, eelgrass has experienced a dramatic decline in both the Virginia and Maryland sections of the Chesapeake Bay.

In Maryland, eelgrass has been found abundantly from Eastern Bay south to Smith Island. During the last six years there has been a significant decline of the Zostera beds in these areas, with most of the decline occurring between 1972 and 1973 (see Table 4, Figure 8). The areas most affected by this decline were Manokin River, Bloodsworth Island, Honga River, Big and Little Annemesex Rivers, Pocomoke Sound and Smith Island. There appeared to be a minor decline in these areas between 1975 and 1976.

Zostera marina is the most widespread species of this genus. It is distributed in both the northern Pacific and the northern Atlantic and is the only seagrass extending into the Arctic Circle. On the Pacific Coast of North America, eelgrass extends from Grantly Harbor, Alaska (65°N; McRoy 1968) to Agiahampo Lagoon in the Gulf of California (26°N. Steinbeck and Picketts 1941). On the Atlantic Coast of North America, eelgrass extends from Hudson Bay, Canada, the

Table 4. Percent of sampling stations showing occurrence of *Zostera marina*, Maryland Chesapeake Bay, 1971-1976[a]

Area Code	River system	1971	1972	1973	1974	1975	1976	Number of stations 71	72	73	74	75	76
1	Elk & Bohemia Rivers	0	0	0	0	0	0	15	16	16	16	16	16
2	Sassafras River	0	0	0	0	0	0	10	10	10	10	10	10
3	Howell & Swan Points	0	0	0	0	0	0	12	6	12	12	12	12
4	Eastern Bay	4.26	11.63	0	0	2.17	0	47	43	47	47	46	45
5	Choptank River	5.00	5.17	0	0	0	0	60	58	57	58	57	56
6	Little Choptank River	5.26	0	0	0	0	0	19	19	19	19	19	19
7	James Island & Honga River	41.18	2.94	0	0	0	0	34	34	34	34	34	34
8	Honga River	26.67	16.67	0	0	3.45	3.45	30	30	30	30	29	29
9	Bloodsworth Is.	20.00	15.91	2.17	9.30	6.98	0	40	44	46	43	43	45
10	Susquehanna Flats	0	0	0	0	0	0	27	37	37	37	36	35
11	Fishing Bay	4.00	4.00	0	0	0	0	25	25	25	25	24	25
12	Nanticoke & Wicomico Rivers	0	0	0	0	0	0	30	30	30	30	30	30
13	Manokin River	33.33	40.00	13.33	20.00	7.14	0	15	15	15	15	14	15
14	Patapsco River	0	0	0	0	-	0	21	20	21	21	-	21
15	Big & Little Annemessex Rivers	60.00	50.00	15.00	31.58	27.28	15.00	20	20	20	19	18	20
16	Gunpowder & Bush River Headwaters	0	0	0	0	-	0	9	8	7	9	-	9
17	Pocomoke Sound (Maryland)	18.18	10.00	4.76	-	15.00	9.09	22	20	21	-	20	22
18	Magothy River	0	0	0	0	-	0	12	12	12	12	-	12
19	Severn River	0	0	0	0	-	0	15	15	15	15	-	13
20	Patuxent River	2.00	0	0	0	0	0	50	47	50	50	47	49
21	Back, Middle & Gunpowder Rivers	0	0	0	0	0	0	22	22	22	22	22	22
22	Curtis & Cover Points	0	0	0	0	0	0	20	19	19	19	6	21
23	South, West & Rhode Rivers	0	0	0	0	0	0	8	10	10	8	8	8
24	Chester River	0	0	0	0	0	0	36	36	34	34	36	35
25	Love & Kent Points	0	0	0	0	0	0	8	8	8	8	8	8
26	Smith Island (Maryland)	29.41	45.45	0	11.76	16.67	5.88	17	11	12	17	17	17
	Total	10.26	7.32	1.11	2.46	3.07	1.11	624	615	629	610	553	628

[a] U.S. Fish and Wildlife Service Migratory Bird and Habitat Research Laboratory, 1976

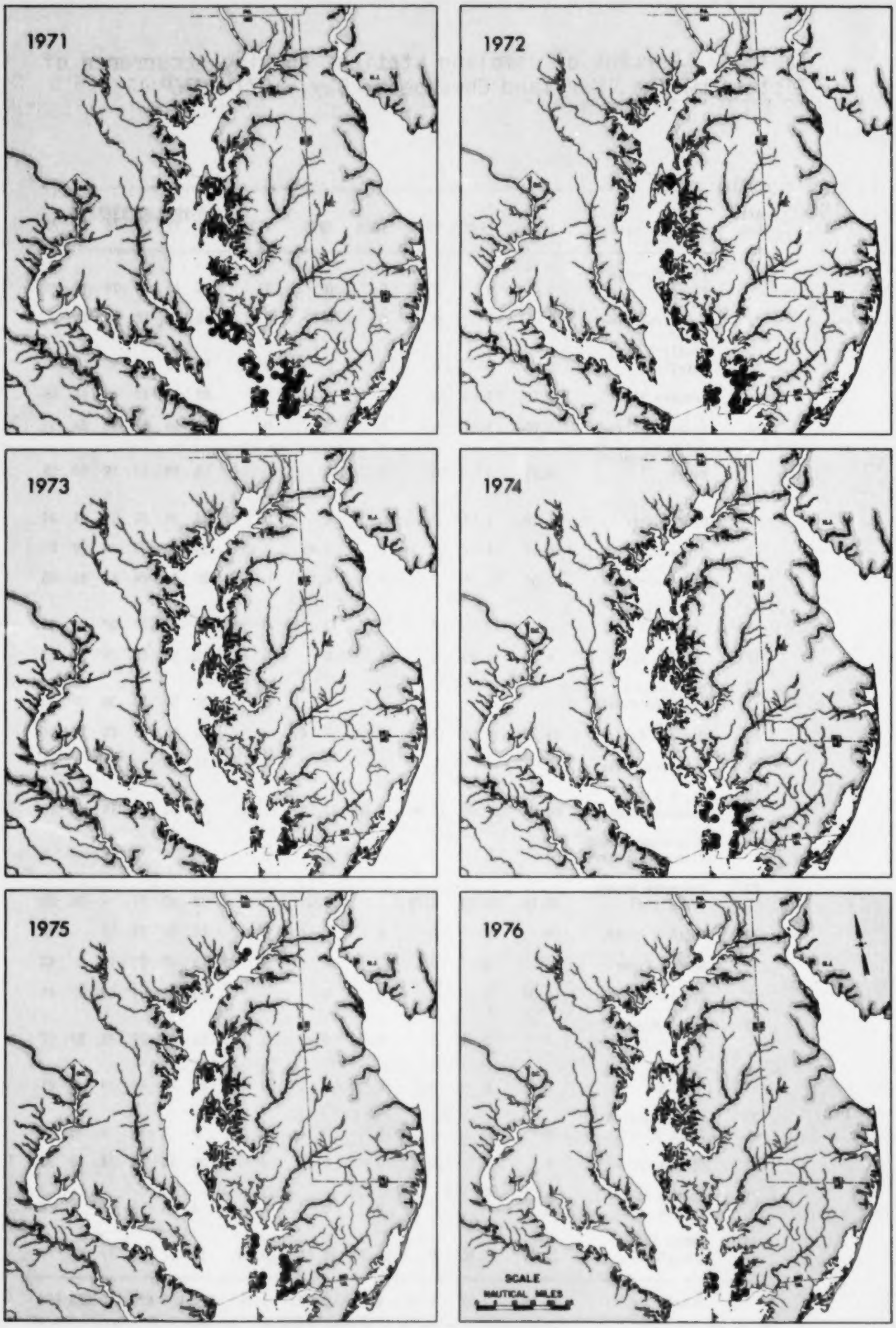

1971
1972
1973
1974
1975
1976
SCALE
NAUTICAL MILES

Figure 8. Distribution of Zostera marina, Maryland Chesapeake Bay, 1971-1976

southern tip of Greenland, and one locality in Iceland (to at least 65°N; Ottam 1934b; Ostenfeld 1918; Phillips 1974a) to Bogue Sound, North Carolina (35°N; Dillon 1971).

Eelgrass populations along the East Coast suffered a dramatic decline in the early 1930s and this decline was attributed to a protozoan parasite, Labyrinthula spp. This decline had a significant effect on the invertebrates and wildlife utilizing eelgrass beds as well as significant effects on bottom sediments. (Urner 1934; Renn 1934, 1935, 1937; Cottam 1935a; Stauffer 1937; Tutin 1938; Dreyer and Castle 1941; Addy and Aylward 1944; Dexter 1944, 1950, 1953; Milne and Milne 1951; Cottam and Munro 1954; Johnson and Sparrow 1961; Pokorny 1967; Burkholder and Doheny 1968).

Fluctuations in eelgrass were recorded in several periods prior to the 1930s along the East Coast (1854, 1889, 1894, 1908, 1913, 1915, 1917, 1920 to 1922), but these periods are not as well documented as the 1930s decline (Cottam 1934a, 1935b). Many of these reports represent local changes but the 1894 decline appeared to be a more widespread phenomenon along the East Coast.

In addition to its occurrence on the North American coasts, eelgrass is present along the European coast from the White Sea (65° to 70°N) and Cheshskaya Guba in northern Russia to Spain, near Gibralter (Zenkevitch 1963; Hartog 1970). It is rare in the Mediterranean, Adriatic and Aegean Seas but is common in the Black Sea. It has been reported from only one locality on the North African coast in Algeria (Hartog 1970).

Along the East Asian coast, eelgrass is common in Japan and Korea but little is known about its distribution in China and the U.S.S.R. (Hartog 1970).

Eelgrass populations along the European coasts also suffered a dramatic decline in the 1930s, again attributed to the protozoan Labyrinthula though other causes were suspected (Cottam 1933a, 1933b; Blackburn 1934; Butcher 1935; Atkins 1938; Wilson 1949; Rasmussen 1973, 1977). Rasmussen (1973, 1977) examined eelgrass fluctuations in the Isefjord area of Denmark from 1899 to 1968, relating temperature fluctuations to disappearance of eelgrass, and concluded that a combination of warm winters and warm summers were responsible for past periods of decline. Kikuchi (1974a, 1974b) found large portions of eelgrass meadows had disappeared in the last 15 years in Japan. These losses were attributed to land filling, water pollution accompanying rapid industrialization of the coastal area and increased turbidity also due to increased development and subsequent runoff. Hartog and Polderman (1975) examined changes in eelgrass populations in the Dutch Waddenzee between 1869 and 1973. They found considerable changes between 1869 and 1930, prior to the "wasting disease" destruction and regarded these changes as normal long-term fluctuations within the large-scale pattern of the dynamic equilibrium of the Waddenzee ecosystem.

The 1930s wasting disease substantiantly reduced the sublittoral populations of eelgrass in the Waddenzee while littoral populations were unaffected. Coincident with the wasting disease was the closure of the Ziderzee, an inland sea formerly in open connection with the Waddenzee. The closure caused considerable hydrological changes, resulting in erosion of substrates formerly occupied by eelgrass. This closure may have aided in the decline but more importantly, prevented successful recruitment in later years because of these new conditions. Fluctuations occurred also between 1932 and 1965, but since 1965 there has been a general decline of eelgrass which was not considered part of the normal fluctuation (Hartog and Polderman 1975). The exact cause for this recent decline had not been ascertained but it was strongly suggested by Hartog and Polderman that increasing pollution was the major factor.

There has been no reported recent decline of eelgrass similar to that found in the Chesapeake Bay, northward of the Bay to Maine. However, there appear to be no programs specifically designed to investigate distribution of eelgrass. Several individuals working on eelgrass in Long Island Sound have not noticed any decline (Churchill, personal communication).

Environmental Factors Affecting Distribution

Temperature. Eelgrass is a perennial plant and exhibits distinct phases of seasonal growth. Setchell (1929) attributed these phases to changes in ambient temperature and delineated five periods of activity governed by 5 C sea water temperature intervals. In the Chesapeake Bay, eelgrass has been observed to grow slowly during the winter months (December through February) as noted by the growth of seedlings during this period (Orth, personal observation).

Biebel and McRoy (1971) studied temperature responses of eelgrass in tidepools (short form) and subtidal areas (long form). They showed that photosynthesis increased with temperature in the tidepool form up to 35 C but in the subtidal form up to 30 C only. Above these temperatures photosynthesis decreased sharply. They also found that both forms withstood temperatures to -6 C but died after exposure to -9 C. Eelgrass has been found in good vegetative condition under Arctic ice (McRoy 1969) which gives some indication of its tolerance to cold conditions.

Thus, it appears that the phenology of eelgrass at different geographical localities may be determined by those climatic factors (mainly temperature) specific to that region.

Salinity. Eelgrass is a euryhaline species and can tolerate salinities ranging from 8 ppt to full strength sea water (Ostenfeld 1908; Tutin 1938; Martin and Uhler 1939; Arasaki 1950a, 1950b; Phillips 1974a). In the York River Virginia, eelgrass does not grow above Clay Bank where salinity does not fall below 10 ppt (Orth 1971).

Biebel and McRoy (1971) studied the salinity tolerance of eelgrass in Alaska and found plasmatic resistance in a range from distilled water up to about 90 ppt. In about 120 ppt, leaf pieces were completely dead within 24 hours. Photosynthesis which had its maximum in normal seawater decreased to nearly zero in distilled water and in about 60 ppt salinity.

Osterhaut (1917) found that eelgrass exposed to alternating fresh and marine water responded differently to fresh water than plants only exposed to marine water. He proposed physiological types of eelgrass, i.e., eelgrass growing at different salinity regimes or exposure to alternating salinities.

However, he found no difference between eelgrass types when roots were exposed to fresh water. Roots of both types were killed after exposure for a few minutes.

Substrate. Eelgrass has been found growing on a wide variety of substrates ranging from pure firm sand to pure soft mud (Phillips 1974a). Within the Chesapeake Bay, eelgrass has been found primarily on sandy substrates (sediments with 70 percent sand or more)(Orth 1971, 1973, 1975a, 1975b, 1977b, in press) though Orth (personal observation) has observed eelgrass growing in finer sediments.

Light, Depth and Turbidity. Eelgrass requires some minimum of sunlight to be able to maintain its photosynthetic machinery. Thus, its vertical distribution will be determined by light penetration, which in turn is affected by depth and turbidity.

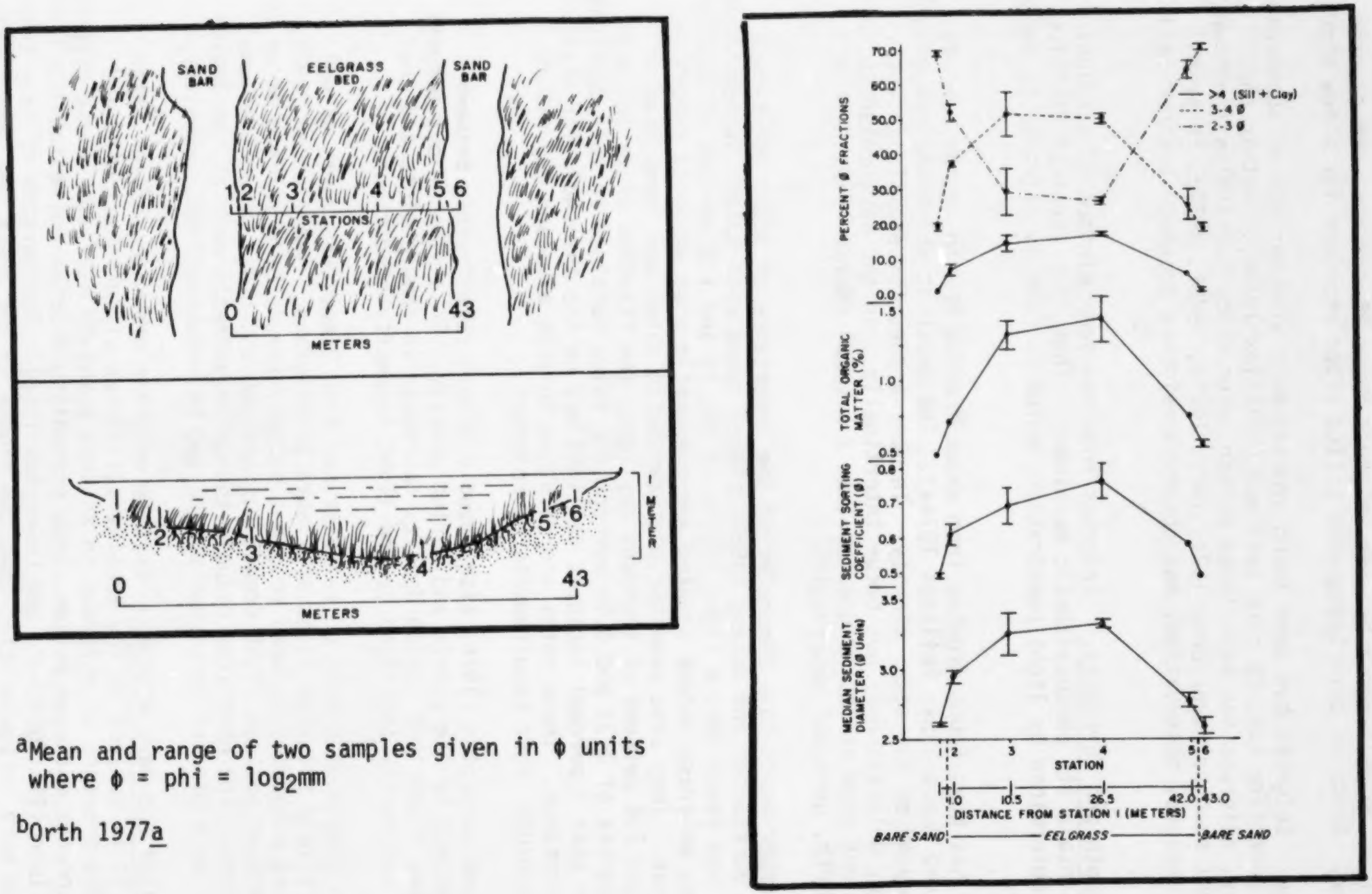
Eelgrass has been found growing from about 2m above MLW to depths down to 30 m (Cottam and Munro 1954; Phillips 1974a). The depth at which maximum growth occurs will depend on the water clarity. In many areas, for example in the Chesapeake Bay, eelgrass does not occur intertidally. In the Bay, eelgrass apparently does not grow at depths greater than 2 m (MLW) (Marsh 1970; Orth and Gordon 1975; Orth, personal observation).

Several experiments have demonstrated the importance of light for growth of eelgrass. Burkholder and Doheny (1968) placed cages with nylon screen shades over eelgrass and found that a light levels of 20, 10 and 1.6 percent of daylight in Long Island, eelgrass became stunted and eventually died while it flourished in control areas. They also examined photosynthetic rates and found that at levels of 10 and 1.6 percent of incident daylight, the fixation of carbon was reduced in eelgrass of 0.33 and 0.14 percent the value obtained in full daylight. They suggested that 1 percent incident daylight may be the threshold for photosynthetic maintenance. These rates are relative to this area and may not represent the absolute light requirements of eelgrass.

Backman and Barilotti (1976) also found a direct relationship between irradiance received by the plants and turion density. In a simple, but elegant experiment, they found that turion density decreased in shaded plots and that flowering was apparently inhibited by low light intensity conditions.

Epiphytic growth on blades of eelgrass can also limit the growth of eelgrass by effectively reducing light available to eelgrass. Sand-Jensen (1975, 1977) found that epiphytes reduced eelgrass photosynthesis by a maximum of up to about 31 percent at optimum light conditions and ambient HCO_3^- concentrations in his study area. The epiphytes reduced the photosynthetic rate of the leaves by acting both as a barrier to carbon uptake and by reducing light intensity.

Current Wind and Wave Action. Because eelgrass grows in shallow water environments, it is subject to the erosive activities of currents, wind and wave action. The presence of eelgrass in an area normally mitigates these effects by baffling currents and wave action, thus preventing erosion and destabilization of the bed (Wilson 1949; Ginsburg and Lowenstam 1958). The margins of many eelgrass beds show more irregularities than the middle of the grass bed, especially



[a]Mean and range of two samples given in ϕ units where ϕ = phi = $\log_2$mm

[b]Orth 1977a

Figure 9. Sediment characteristics across a Zostera marina bed, York River, Virginia[a,b]

in exposed areas, indicating a greater intensity of erosional activity along these edges (Hartog 1970; Orth and Gordon 1975).

Along the bayside of the Delmarva Peninsula, Virginia, the presence of an extensive offshore intertidal sand bar allows the development of eelgrass beds behind the bars, by reducing wave activity from offshore winds. Intertidal sand bars interspersed with eelgrass between the bars are features typical of some exposed areas in the Chesapeake Bay (Orth 1975b; Orth and Gordon 1975) and the persistence of these sand bars is probably related to the hydrodynamics and bottom topography of the region. As with the offshore bar, the presence of these smaller sand bars allows the growth of eelgrass by reducing wave activity.

Current itself apparently is important to the growth of eelgrass. Phillips (1974a) found luxuriant eelgrass where currents reached speeds up to 3.5 knots. Conover (1958, 1968) found similar results for eelgrass in Rhode Island and Massachusetts. He showed that gross and net production rates could be raised or lowered by changing the recirculation velocity of the seawater in experimental tanks. Eelgrass attained peak biomass figures in current speeds of around one knot with a sharp dropoff above this figure. The connection between current velocity and plant metabolic rates is believed to be a function of the rate of transfer of required gaseous and dissolved nutrients from the water into the organisms.

Sedimentation. As mentioned in the previous section, eelgrass has the ability to trap and bind sediments, providing some stability to the bottom substrate. This trapping effect usually results in a slight elevation of the eelgrass bed above the surrounding unvegetated bottom (Hartog 1979; Orth, personal observation). In some areas, movement of sand may be so great, e.g. during periods of intense storm activity, as to actually cover the grass thereby killing it. Blois et al.(1971, cited in Hartog 1970) described this phenomena to occur in one day. However, they observed new rhizome growth in the sanded area resulting in a new horizon of rhizomes. In one *Zostera* bed they found seven separate rhizome layers. In one location in the Chesapeake Bay, Orth (personal observation) found an edge of a sand bar migrating across an eelgrass bed. Recently covered grass that was still green was observed 20 cm or more beneath the surface of the sand.

Some areas in the Bay have characteristic sand bars parallel to the shore with slight depressions between the bars. In some areas, these depressions are inhabited by eelgrass and the trapping and binding properties of eelgrass are evident here also (see Figure 9). Sediments just inside the grass bed are distinctly different from those immediately outside the bed and they get progressively finer from the edge to the center of the bed.

Susceptibility. Because eelgrass lives in an aquatic environment, it is subject to pollutants from both point and non-point sources. Unfortunately, there is very little information concerning the effects of various toxicants on the growth and survival of eelgrass.

Thomas (1967) and Thomas and Duffy (1968) tested the effect of nine different herbicides on the growth of eelgrass (see Table 5). They found that

Table 5. Chemicals tested, concentrations used and main results of eelgrass control experiments[a]

Chemical		Application rates/ha		
Trade name	Chemical name	Formulation	Active ingredient	Results
Reglone	Diquat (0.02 kg/liter)	1.9 - 83.3 liter	0.5 - 20 kg	None
Gramoxone	Paraquat (0.02 kg/liter)	1.9 - 83.3 liter	0.5 - 20 kg	None (some faunal
Drillex	-	796 liter	771 liter	Total mortality of all biote except oysters
Kurosal G	Silvex (23%)	49.0 - 186 kg	11.2 - 42.8 kg	None
Aquathol	Endothal (5%)	195 - 783 kg	36.2 - 157 kg	None
Aquathol Plus	Endothal (5.1%) Silvex (5.6%)	297 - 594 kg	31.8 - 63.6 kg	Less than 1% kill at max. rate
Fenac 5G	Fenac (5.5%)	440 - 710 kg	24.2 - 39.1 kg	None
Aqua-Kleen	2,4-D (29%)	147 - 440 kg	42.8 - 127.6 kg	50-100% at max. rate; none at min. rate
Casoron G4	Dichlorobenil (4%)	187 - 550 kg	7.4 - 22.1 kg	50-90% at max. rate; 25% at min. rate
Aqua-Kleen	2,4-D (29%)	223 - 446 kg	45 - 89 kg	90-100%
Casoron G4	Dichlorobenil (4%)	200 - 446 kg	7.7 - 17.6 kg	None

[a] Thomas 1967

only dichlobenil and 2,4-D had any effect on eelgrass and further testing showed 2,4-D to be extremely toxic. Areas killed by 2,4-D ranged from 3.2 to 5.8 times the area treated because of current-born herbicide.

Nutrient Response. Nutrient cycles in eelgrass beds are complex and the problems associated with nutrient limitation, sites and rates of uptake still need further investigation. Recent work has shown that eelgrass can absorb phosphorus and nitrogen across the leaves and roots and transport it to other parts of the plant (McRoy and Barsdate 1970; McRoy et al. 1972; McRoy et al. 1973; Barsdate et al. 1974; Penhale 1976). Eelgrass also serves as a "nutrient pump", pumping phosphate and ammonia from the sediments through the plant and releasing them into the water. The flow of phosphorus can occur in the opposite direction, but the net effect is to transport phosphorus from the sediment to the water, effectively increasing the concentration of these nutrients in the water.

This transport of nutrients is very important to epiphytic algae because they apparently utilize carbon, nitrogen and phosphorus excreted by the plant (Harlin 1971; McRoy and Goering 1974; Penhale 1976). This would potentially allow greater growths of algae on grass blades in nutrient poor water than otherwise possible.

Little work has been conducted with nutrient enrichment studies. Raymont (1947) and Marshall and Orr (1948) studied the effects of the addition of fertilizers (sodium nitrate and superphosphate) to sea lochs, primarly to study the effects on algal primary productivity and resultant fish production. They noticed an increased abundance of eelgrass but no actual measurements were made of this increase, nor were appropriate controls available (i.e. they did not differentiate normal seasonal growth from growth presumably enhanced by fertilizing). Marshall and Orr suspected that eelgrass utilized the added nutrients before the phytoplankton. Orth (in press) working in a more controlled experiment off Hungars Creek, Delmarva Peninsula, found that the biomass, length and number of vegetative turions of eelgrass increased significantly when commercial fertilizer was added to sediments in an eelgrass bed (see Figure 10). Fertilizers with N-P-K values of 5-10-10 and 10-10-10 were used.

Eelgrass appears to be very effective in concentrating various elements from the water (e.g. major elements such as carbon, nitrogen and phosphorus, minor elements such as manganese, aluminum and iron, and trace elements such as copper, cobalt and beryllium) including radioactive isotopes (McRoy 1970b; Barsdate et al. 1974). The accumulation of elements in eelgrass is proportional to the abundance of an element in the water. Thus, it would appear that any increase in a major element would be reflected in eelgrass.

Productivity

Table 6 summarizes the available information on productivity measurements for eelgrass. Data on standing stock (g/m^2 dry weight) and productivity ($g\ C/m^2/day$) of eelgrass at different localities vary widely, ranging 1 to 5157 and 0.04 to 8, respectively. Turion density is also highly variable. The ranges found in Table 6 for those parameters probably correlate with the diverse morphology observed for eelgrass in environmentally different yet often geographically close areas. Considerable attention is necessary when comparing these

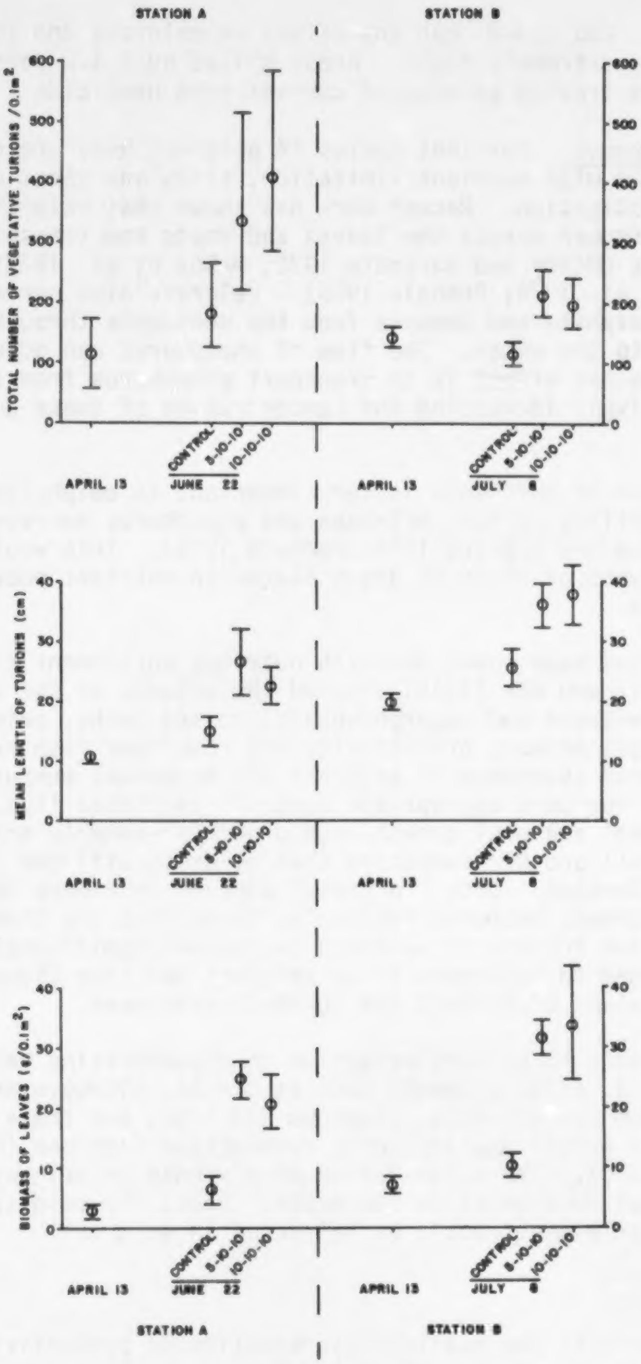


Figure 10. Effect of the addition of fertilizers 5-10-10 and 10-10-10 on the growth of Zostera marina

Location	Standing stock (g dry/m^2)			Density (no./m^2)	Productivity		Reference
	Leaves		Rhizomes + roots		(gC/m^2/day)	(mgC/g dry/hr)	
	Range	Mean	Range				
Denmark[c]	210-487	-	-	-	-	-	Grøntved 1958
Denmark	272-960	-	-	-	2-7.3	(0.27-0.99)	Petersen 1913
Denmark	60-255[e]	-	100-220[e]	1000-1800[e]	-	-	Sand-Jensen 1975
England	-	120	-	-	-	-	Burton 1961
Canary Islands[a]	70-89	-	-	-	0.35	(0.33)	Johnston 1969
Nova Scotia[c]	-	1018	-	-	-	-	Mann 1972
Massachusetts[a]	5-29	-	-	-	0.04	(0.15)	Conover 1958
Rhode Island[b]	5-435[e]	225	-	-	0.04-2.9	(0.15-1.1)	Conover 1968
Rhode Island[c]	-	66.0	-	-	2.6	(2.2)	Nixon & Oviatt 1972
New York[b]	72-2465	438.0	-	-	-	-	Burkholder & Doheny 1968
New York	224-5157	1906	-	-	-	-	Wilson & Brenowitz 1966
New Jersey	153-593	-	-	-	-	-	Moeller 1964
Chesapeake Bay, Va.[b]	6-255	65.8	13-243	390-3140	-	-	Orth (unpublished)
Chesapeake Bay, Va.[c,d]	99-597	277	137-365	1280-5400	-	-	Orth (in press)
Chesapeake Bay, York River, Va.[b]	8-208[e]	-	-	180-780[e]	-	-	Marsh 1970
North Carolina[b]	82-156	-	-	-	0.2-1.2	-	Williams 1966
North Carolina[b]	175-545	273	-	-	0.5-1.7	0.17-2.2	Dillon 1971
North Carolina[b]	-	80.3	-	-	0.9	-	Penhale 1977
North Carolina[b]	1.6-153	-	-	-	-	-	Thayer et al. 1975
North Carolina[b]	1.5-270[e]	-	-	-	-	-	Adams 1976a
California[c]	39-420	240	-	145-470	-	-	Keller & Harris 1966
California	6-192	-	-	-	-	-	Waddell 1964
Washington[b]	1-420	-	5-527	3-1571	0.7-4	(0.33-1.9)	Phillips 1972
Alaska[c]	34-648	-	1-255	710-2101	8	0.55	McRoy 1966
Alaska[c]	62-1840	1000	120-400	599-4576	3.3-3.8	0.33-0.4	McRoy 1970a, 1970b
Japan	70-235	-	-	-	-	-	Kita & Harada 1962
Sakhalin	31-900	900	-	-	-	-	Shchapova & Vozzhinsekaya 1960
White Sea	164-203	-	-	-	-	-	Kuznetsov & Marveeva 1963
Black Sea	100-150	-	-	-	-	-	Zenkevitch 1963

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[a] Modified and corrected from McRoy and McMillan (1977)
[b] Seasonal or monthly samples
[c] Summer sampling or does not represent samples from all seasons
[d] Eelgrass enriched with commercial fertilizer
[e] Interpolated from graphs

studies; some numbers represent samples from seasonal studies, where the range of values will be greater than those from studies where samples were only collected during the season of maximal biomass. Sampling techniques vary widely, especially in productivity studies, and make it difficult to draw distinct conclusions.

Eelgrass is a temperate seagrass and has a distinct growing season. Maximal biomass of leaves is normally attained from June through August (Penhale (1977 found this in March) while minimal biomass is attained during January and February.

Consumer Utilization

Despite the abundance of eelgrass along the coast, there are relatively few animals that directly utilize the grass as a food course. However, this fact alone does not belie the importance of eelgrass as a habitat and nursery for many forms of invertebrates and vertebrates, both juveniles and adults. These organisms often undoubtedly serve as a source of food of species at higher trophic levels.

The fauna and flora of eelgrass beds can be divided into several categories: the epibiota (animals and plants utilizing the blades of eelgrass as a substrate), the infauna, (animals found on or in the sediments) and the motile invertebrates and vertebrates. Within this last group there can be subgroups such as permanent residents, seasonal residents, transients which utilize a wider area than the seagrass bed and occasional migrants.

The epibiota and infauna represent a diverse and complex assemblage which includes micro- and macroalgae, protozoans, hydrozoans, anthozoans, turbellarians, gastropods, isopods, amphipods, polychaetes, oligochaetes, bivalves, decapods and barnacles. Many of these groups exhibit distinct seasonal pulses of abundance depending on their individual spawning periods (Barnard 1970; Kikuchi 1968; Kikuchi and Peres 1977; Lappalainen 1973; Lappalainen and Kangas 1975; Levinton 1977; Levinton and Bambach 1975; Marsh 1970, 1973, 1976; Nagle 1968; Orth 1971, 1973, 1975a, 1975b, 1977a,b; Rasmussen 1973; Thayer et al. 1975).

The biotic community within grass beds is quite distinct from the communities in adjacent unvegetated areas. Because of the lack of suitable substratum, there is usually very little epifauna in bare sand or mud areas. The animals are primarily using blades as a substratum or in the case of herbivorous gastropods, grazing on the microalgae that grow on the blades. In experiments where artifical grasses were used, an epifaunal community similar to that found on the real seagrass developed on the artificial seagrass (Moulton 1971; Orth, unpublished data).

The infaunal community is also quite distinct from adjacent unvegetated areas. There is a tremendous increase in numbers of species and individuals in grass areas and this may be related to increased sediment stability, microhabitat complexity of food supply (Kikuchi 1966; Williams and Thomas 1967; Nilsson 1969; Adams and Angelovic 1970; Tenore 1975; Orth 1975b, 1977b; Thayer et al. 1975). Orth (1975b, 1977b) working with infauna of Chesapeake Bay eelgrass, found infauna to increase in density and diversity from the edge of an eelgrass bed to the center of the bed (see Figure 11) and also with increasing size of the bed (see Figure 12). He related this increase to the sediment stability component of eelgrass and showed that decreasing the stability of sediments experimentally (removing blades of grass by clipping, simulating wave action) and naturally (cownose ray activity (Orth 1975a)), decreased the density and diversity of the infauna.

The motile community is also diverse and quite distinct from surrounding unvegetated areas (Orth and Heck, unpublished; Briggs and O'Conner 1971; Kikuchi 1974b). Utilization of these grass areas by fish and invertebrates may be for distinctly different reasons. Hardwick (1973) found that on the West Coast, the herring (Clupea harengus pallasi) used eelgrass leaves to lay most of their eggs. Juveniles and adults of many species may utilize eelgrass for protection. The blue crab (Callinectes sapidus) is found in greater abundance in eelgrass both as juveniles and adults (Lippson 1970; Sulkin 1973, 1977; Orth and Heck, unpublished). Many species may use eelgrass beds primarily as a habitat and feeding ground because of the abundance of food. Barry (1974) found shrimp, (Hippolyte californiensis) which inhabit eelgrass beds on the West Coast, were able to recognize the leaves of eelgrass by discriminating it on the basis of its form.

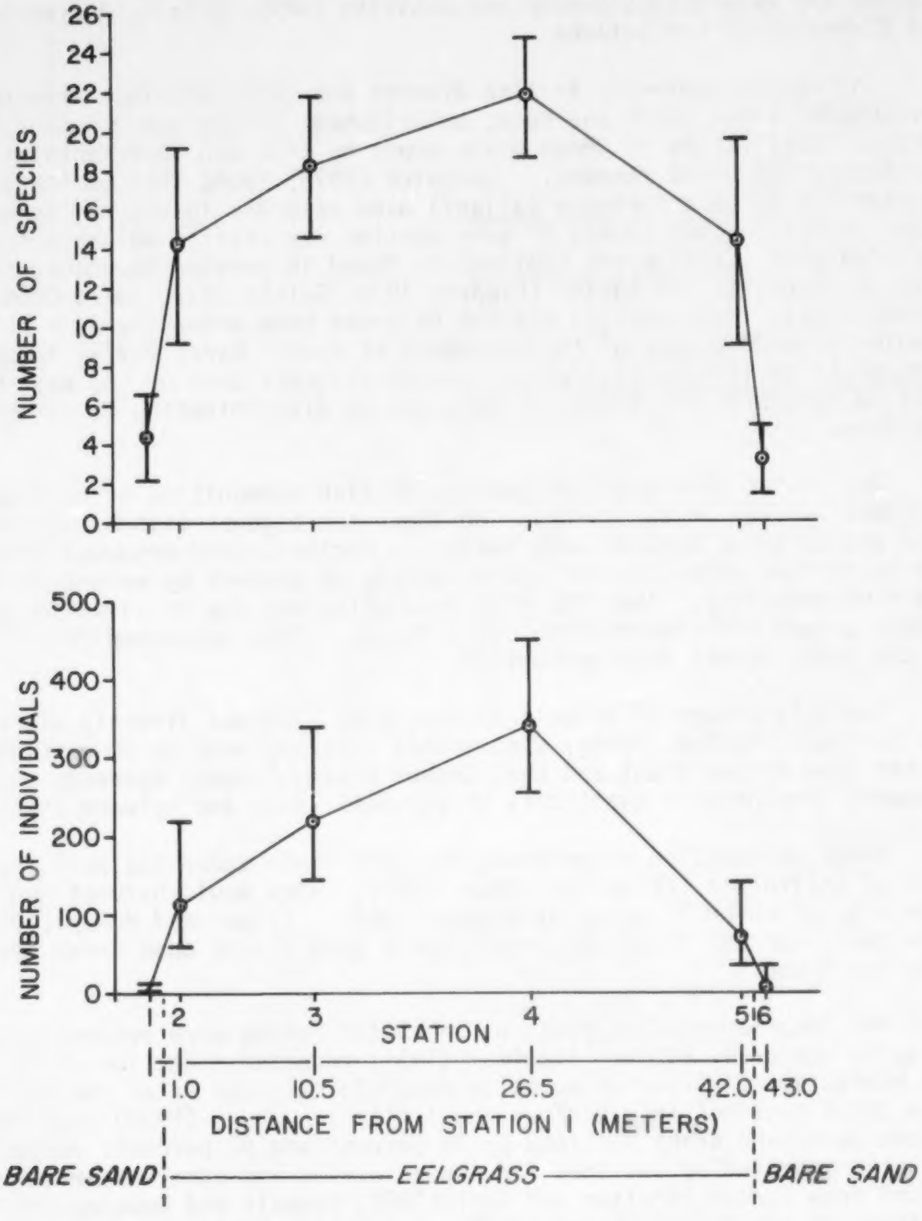
One of the more complete studies of fish communities of eelgrass was done by Adams (1976a, 1976b, 1976c). He found the highest fish biomass when temperature and eelgrass biomass were maximal. Further, food produced within the grass bed could have accounted for approximately 56 percent by weight of the diet of the fish community. The high fish production was due to juveniles which had higher growth efficiences than older fishes. They accounted for 79 to 84 percent of the total annual fish production.

The only groups of animals that consume eelgrass directly are waterfowl and sea turtles. Cottam (1934b) stated that eelgrass made up 80 percent of the winter food of Sea Brant and that Canada Geese, Scaups, Redheads and Black ducks consumed considerable quantities of eelgrass (Addy and Aylward 1944).

Human consumption of eelgrass has been found among the Seri Indians in the Gulf of California (Felger and Moser 1973). They would harvest the seed and prepare a gruel which is eaten with other food. Felger and McRoy (1975) made bread from the flour and found the bread "had a good flavor when fresh, and is somewhat like rye bread."

The importance of eelgrass as a habitat became more evident after its decline in the early 1930s. Stauffer (1937) reported a decline of 33 percent of the characteristic species normally found in eelgrass after the demise. There were sharp declines in waterfowl populations. Urner (1934) reported that Canada Geese and Brant declined by 75 percent and 92 percent, respectively, in New Jersey. Similar declines for these species and other waterfowl were reported from Europe (Bruijns and Tanis 1955; Ranwell and Downing 1959) and other sections of the United States (Moffit and Cottam 1941).

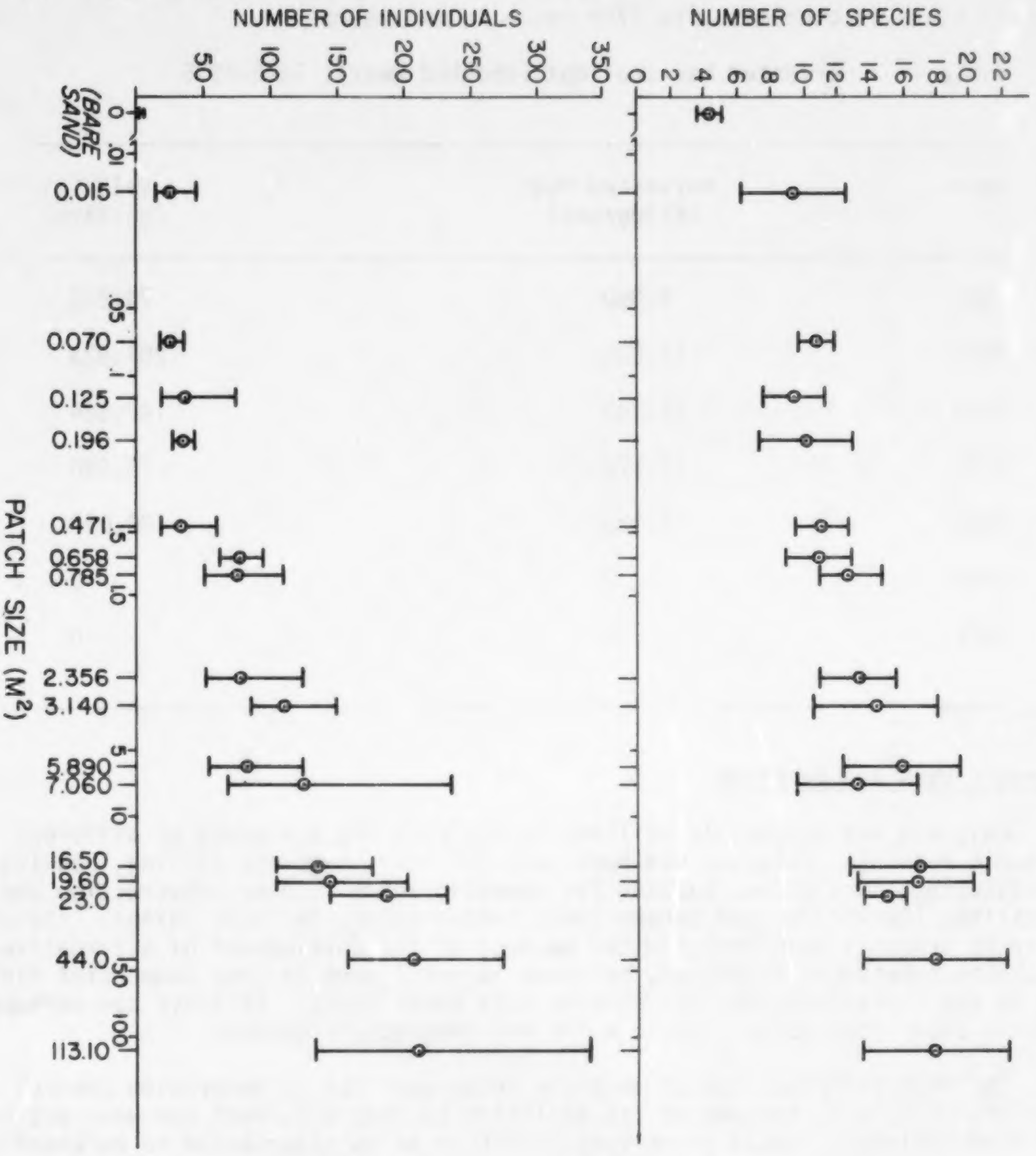
Another serious effect was on the bay scallop (Aequipecten irradians) population on the Eastern Shore of Virginia. The commercial fishery that amounted to a harvest of over 14,000 kg/year in the late 1920s and early 1930s was reduced



[a]See figure 9 for station locations

[b]Orth 1977b

Figure 11. Faunal characteristics across a Zostera marina bed[a,b]



[a]Mean and 95 percent confidence levels

[b]Orth 1977b

Figure 12. Faunal characteristics of a _Zostera_ _marina_ bed[a,b]

to nothing by 1933 (see Table 7). The species still has not returned to this area in marketable quantities. The life cycle of the bay scallop depends upon the presence of eelgrass as a setting substrate for the post-veliger larvae (Gutsell 1930). Without this substrate, the scallop population would decline rapidly since it completes its life cycle in two years.

Table 7. Harvested bay scallops (shucked meat), 1925-1975

Year	Harvested Meat (Kilograms)	Value (dollars)
1925	5,050	74,272
1929	16,038	207,883
1930	25,549	147,564
1931	17,170	78,990
1932	9,220	80,090
1933	0	0
1975	0	0

Economic Uses and Problems

Eelgrass was apparently utilized in the past for a variety of different economic reasons. Eelgrass had been used for fuel, mattress filling, packing, upholstering, insulation, bedding for domestic animals, dike construction and fertilizer (Burkholder and Doheny 1968; Cottam 1934b; Phillips 1974a). Its use today is probably much more limited because of the development of alternative synthetic substances. However, eelgrass is still used by some commercial fishermen in the Chesapeake Bay for packing soft shell crabs. Eelgrass can perhaps be best used today as an organic mulch and compost for gardens.

The most important use of eelgrass today would be as an erosion control barrier. Eelgrass, because of its abilities to reduce current and wave action and bind sediments, could prove very effective as an alternative to bulkheading. However, there have been no published attempts to establish eelgrass to stabilize the shoreline. One would first have to establish the cost-effectiveness of planting eelgrass or establishing a bulkhead prior to any attempts to initiate this activity.

Some undesirable features of eelgrass are hazards to small boats, hindrances to navigation and interference with water skiing and swimming. On the Pacific coast, eelgrass interfered with oyster harvesting and decreased amount of useable bottom for oyster culture. Eelgrass was considered a nuisance and measures were tried to eradicate it (Thomas 1967; Thomas and Duffy 1968).

Thus, today, there are very few direct uses of eelgrass. However, the indirect benefits to man, by supplying a food source to commercially important species as the blue crab and sport fishes such as speckled and grey trout and acting as a natural erosion control barrier, make this community very valuable in the Bay.

Man's Impact on the Eelgrass Ecosystem

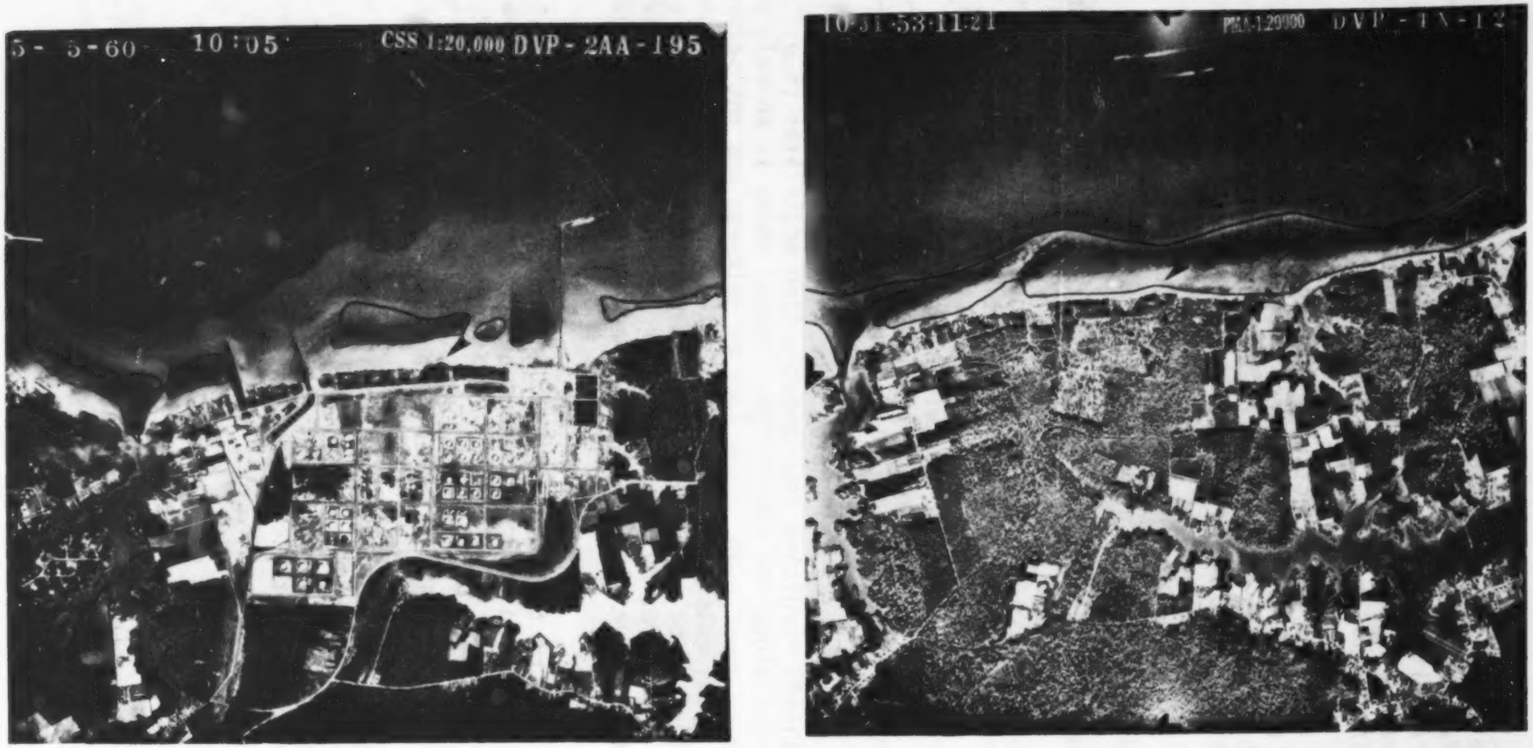
Because eelgrass inhabits the shallows of the coast zone, the increasing use of this zone by man for industrial and recreational activities and the coastal zone's susceptibility to point and non-point sources of pollution, make this species extremely vulnerable to potential degradation.

Several examples from the Chesapeake Bay will illustrate these points. Figure 13 shows an area of the south shore of the lower York River, Virginia, taken in 1953 and 1960. The extent of the eelgrass bed is delineated in the outlined area. The construction of an oil refinery and oil-generated power plant in the seven years between the two photographs resulted in the extensive loss of eelgrass, most likely resulting from construction activities (i.e. increasing water turbidity) along the shore line. Today, there is no eelgrass in front of the refinery or power plant. The loss of eelgrass was not natural as eelgrass increasedin areal extent directly across the river during this same period. Recolonization of eelgrass after construction was finished may have been prevented by alteration of sediments or perhaps the heated effluent from the power plant may have altered normal temperature patterns around this area.

Figure 14 shows an area of the Severn River, a tributary of the Mobjack Bay, Virginia. The light colored area surrounded by the darker band (eelgrass) was the result of a dredging operation by a private individual for fill material. This operation eliminated 1.3 ha of eelgrass. The speckled appearance of this area is due to small patches of eelgrass and widgeongrass colonizing the area.

Figure 15 shows a portion of an eelgrass bed in the Mobjack Bay, Virginia. The light bands crossing the eelgrass bed are the result of intense boating activity in the grass area. This is not an atypical situation as many eelgrass beds in the lower Bay, especially those in shallower areas, exhibit this crisscrossing pattern.

Dredge and fill operations and spoil disposal pose a serious threat to eelgrass survival. These activities alone can destroy or reduce eelgrass abundance but the resultant activity usually increases the turbidity of a region and this too may cause a decrease in eelgrass abundance, either through silting and smothering or a reduction in light intensity below that necessary for photosynthetic maintenance.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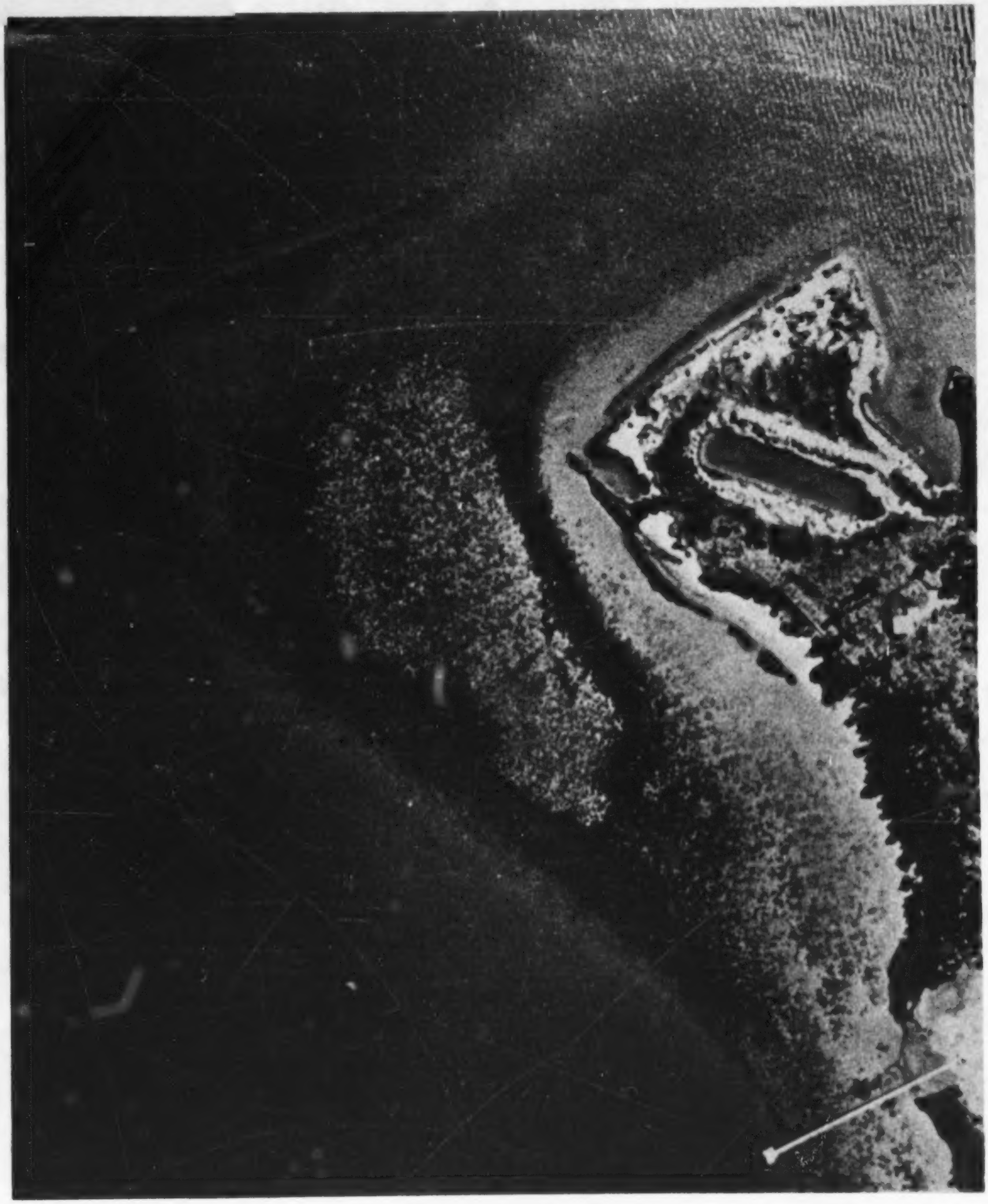
[a]Aerial photographs showing the loss of eelgrass (outlined areas) after the construction of an oil refinery and power plant

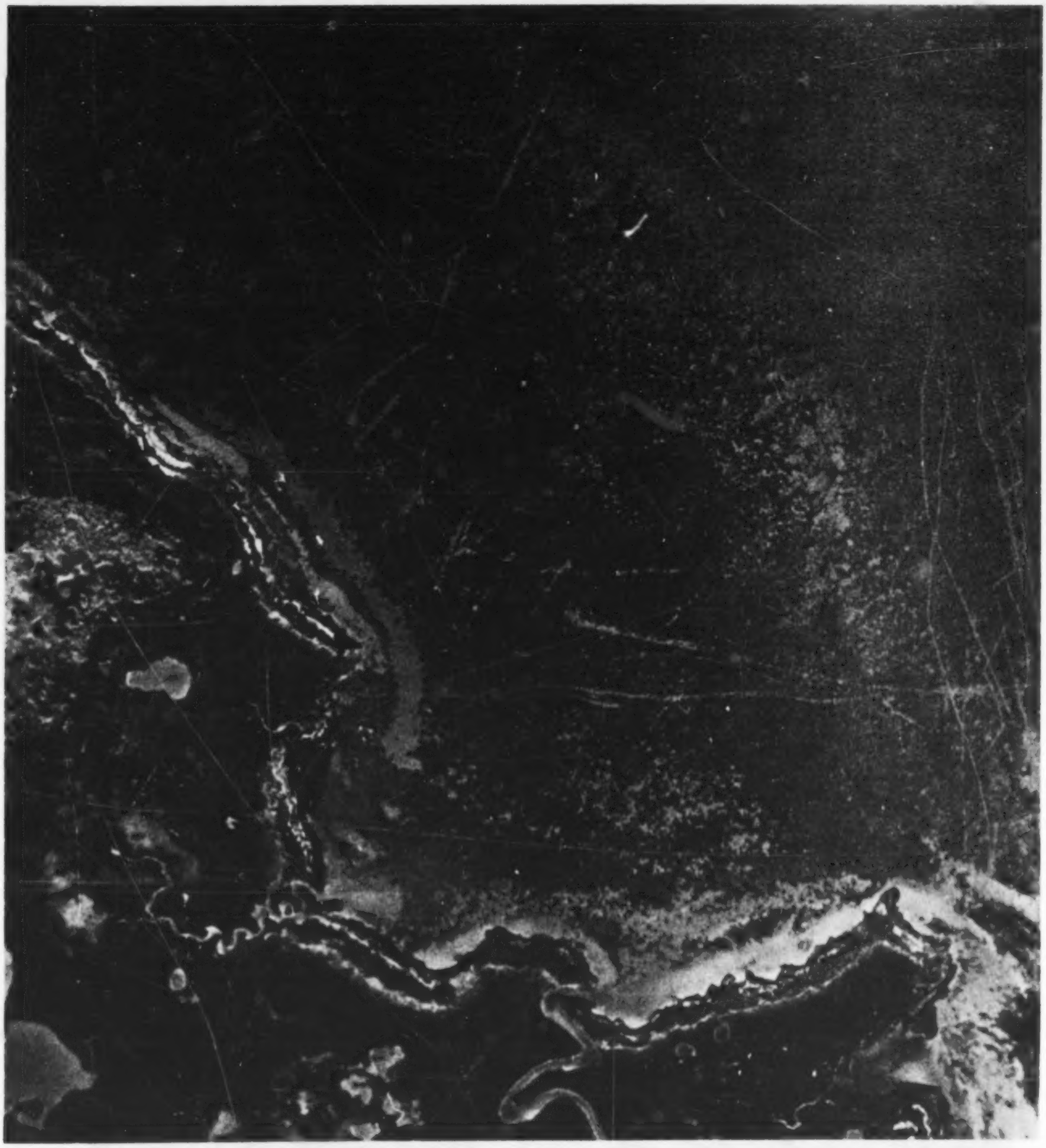
[b] ↗ = reference points for identification

Figure 13. Lower York River, Virginia, 1953 and 1960[a,b]



[a] The light colored area in the middle of the eelgrass bed was the result of dredging activities in this area for fill material for the adjacent marsh.

Figure 14. Aerial photograph of an area in the Severn River, Virginia[a]



[a] Light colored streaks across the eelgrass bed are boat tracks

Figure 15. Aerial photograph of a portion of an eelgrass bed in Mobjack Bay, Virginia[a]

The activities mentioned above in the Chesapeake Bay are not unique to this area. The increasing demand for waterfront property with deep water piers and recreational pressure in the shallows make it imperative for the development of some management scheme for the protection of eelgrass.

Causes for the Decline of Eelgrass in the Chesapeake Bay. Eelgrass in the past has undergone major fluctutations in abundance, the most well-documented being the one that occurred in the 1930s along the East Coast of the U.S. and the West Coast of Europe. The cause for this decline was attributed to a protozoan, *Labyrinthula* (Renn 1934, 1935, 1937) though many other causes were suggested (e.g. salinity, climatical changes, pollution). That *Labyrinthula* was the sole cause for this decline has not been proven conclusively and recent papers (Rasmussen 1973, 1977) cite this as a secondary agent because *Labyrinthula* has been found living on healthy eelgrass instead of being only on dead eelgrass.

More recently, temperature has been implicated as the major factor for causing changes in eelgrass abundance (McRoy 1966; Rasmussen 1973, 1977; Orth 1976). Temperature is extremely important in regulating the growth, reproduction and senescense of eelgrass and it is not an unreasonable assumption to implicate temperature as a primary agent in the decline of eelgrass.

Rasmussen (1973, 1977) examined patterns of abundance of eelgrass in Denmark and correlated this with annual temperature patterns. He attributed eelgrass fluctuations to warm summers and exceptionally mild winters. He postulated that high water temperature would weaken the plant, which directly affected metabolism and indrectly weakened the plant making it more susceptible to the ever-present bacteria and protozoa.

The first recently recorded decline of the major eelgrass beds in the Bay occurred in the early fall, 1973, and another mass defoliation in the early fall of 1975. Some eelgrass areas, for example in Mobjack Bay, did not decline during 1973 but declined in 1975. There were reports of declines of eelgrass in 1972 after Agnes but these were limited to the up-estuary limits of eelgrass. It appears that the freshet produced by Agnes did not cause the decline unless there was a delayed reaction which did not appear until a year later. This appears unreasonable since eelgrass declined almost 14 months after Agnes.

The period of the early to mid 1970s was marked by a warming trend characterized by warm winters (see Figure 16). Because eelgrass is approaching its southern limits in the Chesapeake Bay (North Carolina is the southern limit of eelgrass on the East Coast) any shift in the critical temperature regime may dramatically affect the growth cycle of eelgrass. There is no documented evidence from North Carolina to indicate recent changes in abundance of eelgrass aside from small areas. However, Dillon (1971) found the almost complete die-off of eelgrass in the late summer with subsequent regrowth from seedlings and rhizomes. Whether this phenomenon observed in North Carolina is an annual occurrence is unknown, but this was similar to what had happended in the Bay, except that in many areas, e.g. the York River and the Rappahannock River, there has been very little regrowth.

There have been no reports of eelgrass die-backs north of Chesapeake Bay to Maine but there have been no extensive surveys of eelgrass populations in

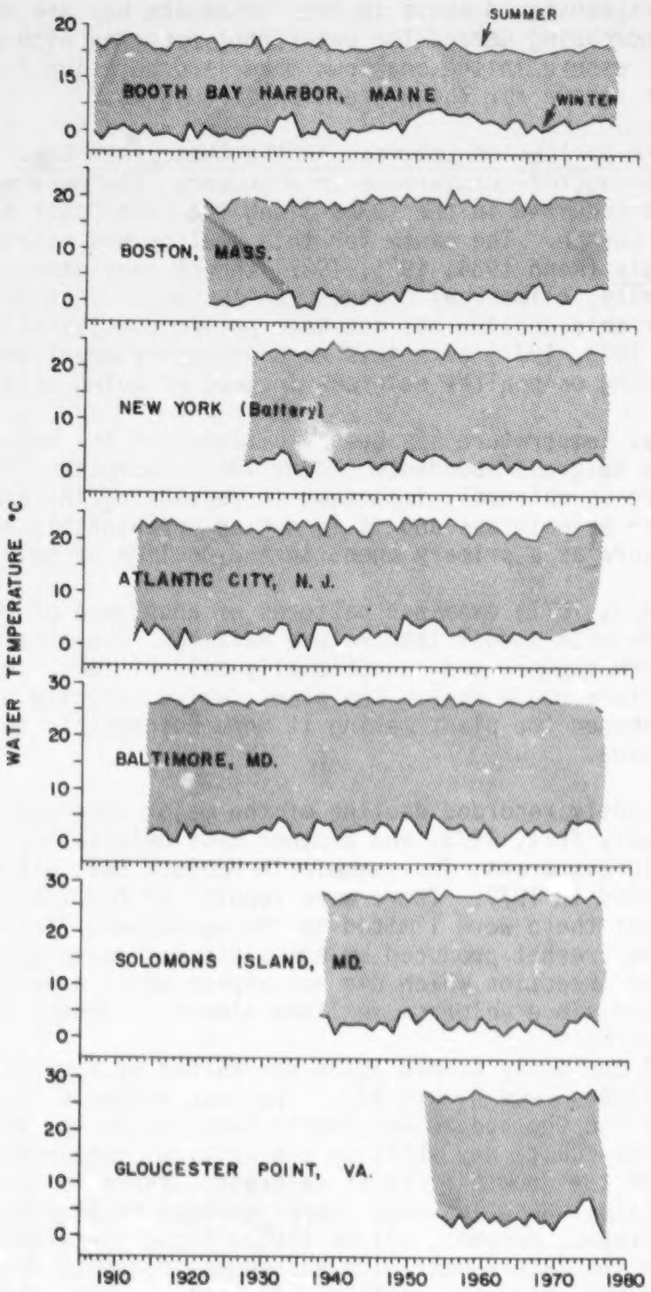


Figure 16. Mean temperatures for coldest and warmest months at various East Coast locations, 1906-1977

these areas, as have been done in the Bay. An examination of temperature records for various locations along the East Coast (see Figure 16) also indicate a warming trend in the 1970s but with no apparent decline of eelgrass.

The temperature records for the East Coast showed a warm period (i.e. warm winters) around 1932 (see Figure 16). The winter of 1932 was unusual because the coldest month was March contrasted to January and February in other years. This period was also preceded by a decade of stable cold winters. If the hypothesis that extreme temperature conditions stress the plants is correct, then extreme variations in certain temperature patterns may account for the eelgrass decline in the 1930s. However, the picture is confused by another warming period in the early 1950s when, supposedly, eelgrass was exhibiting an upswing in abundance. Eelgrass was abundant in 1953 in the lower York River, Virginia, but it is not known what preceded this period in terms of eelgrass coverage. Also lacking are precise data as to rate of recovery and any seasonal changes in already existing beds during periods of recovery.

Temperature may be the primary cause for the disappearance of eelgrass but the lack of controlled experiments over extended periods along with accurate information of present status of existing eelgrass, makes this only an hypothesis for the disappearance of eelgrass.

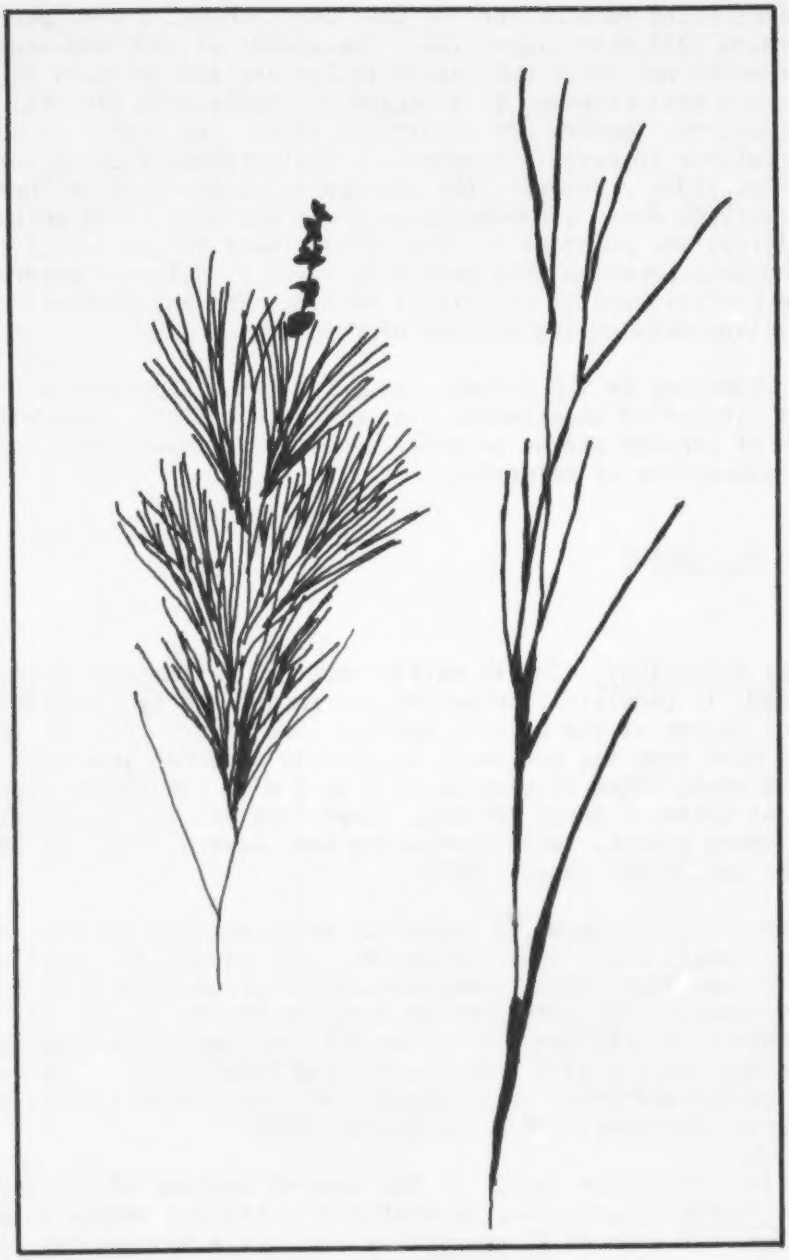
POTAMOGETON PECTINATUS

Biology

General Morphology. In its native setting, *Potamogeton pectinatus*, or sago pondweed, is completely submerged and recognized by a characteristic fanlike spreading of leaves at the water's surface (see Figure 17). *P. pectinatus* derives its name from the pectinate or closely inserted growth of its stems and leaves. The stems often develop up to 2 to 5 m in length and support linear-filiform leaf blades 5 to 35 cm long. Leaf tips are sharply pointed to gradually tapered in young plants. With increasing age, several types of leaf ends can occur on the same plant (Mason 1969).

Two morphological forms of reproductive structures include leafy outgrowths or auxillary tubers and slender peduncles. The peduncles, ranging in length from 50 to 25 cm, have spikes conspicuously interrupted by 2 to 6 whorls, with the axis of the spike becoming lax at maturity (Mason 1969). The starch containing druplets of this species can be distinquished from other pondweeds by the rounded apex of the trap door (Martin and Uhler 1939). The rootstocks are long and straight and often bear rhizoids or specialized turions or winter buds rich in starch (Hotchkiss 1967; Sculthorpe 1967).

There is a reduction series in the general anatomy of Potamogetonaceae with *P. pectinatus* illustrating the most simplistic and reduce form. Unlike other species, the stem of *P. pectinatus* displays a homogeneous zone of phloem surrounding the axial lacuna (Sculthorpe 1967). In the roots the xylem is a single central vessel with the five protoxylem elements absent at maturity (Arbor 1920).



(copied from Hotchkiss 1967)

Figure 17. Sago pondweed (Potamogeton pectinatus)

Reproduction. Potamogeton pectinatus reproduces both sexually and asexually. Sexual reproduction occurs through the production of monoecious flowers arranged in a spike (Yeo 1965a). In a series of studies on reproduction in an irrigation ditch in Huntley, Montana, Yeo (1965a) determined that pollination, fertilization and fruit development occur at the water/air interface. Pollen floats on the surface (Sculthorpe 1967) and after fertilization a druplet is formed. These seeds remain on the rachis of the inflorescence until late fall or until they are removed by moving water (Yeo 1965a).

Vegetative reproduction occurs by two separate mechanisms to form propagules. Subterranean tubers develop from specialized tissue at the branch ends. Occuring in chains of up to five tubers, these structures store large amounts of starch and persist over winter to produce shoots the following spring. A second mode of vegetative reproduction occurs through the development of axillary tubers which form in the fall from the ends of leaf shoots in the leaf axils. Similar to the subterranean tubers, these axillary structures occur singly or in pairs, sink to the bottom and form new plants in the spring (Yeo 1965a).

Based on single season experiments using different sized pools, Yeo (1965b) found that the number of tubers and seeds produced from plants grown from tubers increased as pool sizes increased. In the largest pool (about 23.6 m^2), one sago pondweed plant developed over 36,000 subterranean tubers, 800 axillary tubers and 6,000 seeds within six months. Yeo's experiments readily demonstrated the rapidity with which sago pondweed can colonize a favorable habitat within a short period of time.

Distribution

Potamogeton pectinatus is reported in similar habitats all over the world. Not only does the range of this perennial include fresh streams and ponds, but also brackish coastal waters of the United States and portions of Canada. It has been observed in Russia, India, South Africa, Hungary, Pakistan, and East Germany and exhibits a wide range in altitude from sea level in England to about 5,000 m in Venezuela and Tibet (Klokov and Zimbalevskaya 1974; Misra 1972; Hill et al. 1975; Andrikovics 1973; Ali 1973; Krausch 1976; Ascherson and Grabener 1907.

Though most abundant in the northwestern states and the Chesapeake Bay in the United States (Martin and Uhler 1939), it is reported to be a pest species often encountered in the operation of irrigation systems in the west (Hodgson and Otto 1963) and in water supply ditches in the cranberry bogs of Massachusetts (Devlin 1973).

The continuing survey of submerged aquatic vegetation that is being conducted annually by MBHRL has documented P. pectinatus in the Chesapeake from 1971 to 1977 with the exception of 1975 (see Table 8 and Figure 18). Results of the summer survey for 1977 are not available at this time for the individual species. The 1972 sampling yielded the highest percentage of vegetated stations despite the fact that sampling was conducted subsequent to Hurricane Agnes. MBHRL survey crews found sago pondweed at 5.69 percent of the 615 stations sampled in 1972 from the Chester River south into Tangier Sound on the Eastern Shore and in the Severn and Patapsco Rivers on the Western Shore. In 1973, P. pectinatus

Table 8. Percent of sampling stations showing occurrence of *Potamogeton pectinatus*, Maryland Chesapeake Bay, 1971-1976[a]

Area Code	River system	1971	1972	1973	1974	1975	1976	Number of stations					
								71	72	73	74	75	76
1	Elk & Bohemia Rivers	0	0	0	0	0	0	15	16	16	16	16	16
2	Sassafras River	0	0	0	0	0	0	10	10	10	10	10	10
3	Howell & Swan Points	0	0	0	0	0	0	12	6	12	12	12	12
4	Eastern Bay	0	9.30	6.38	2.13	0	2.22	47	43	47	47	46	45
5	Choptank River	1.67	3.45	7.02	0	0	0	60	58	57	58	57	56
6	Little Choptank River	5.26	10.53	0	0	0	0	19	19	19	19	19	19
7	James Island & Honga River	0	20.59	0	0	0	0	34	34	34	34	34	34
8	Honga River	0	10.00	3.33	0	0	0	30	30	30	30	29	29
9	Bloodsworth Is.	2.50	6.82	6.52	0	0	0	40	44	46	43	43	45
10	Susquehanna Flats	0	0	0	0	0	0	27	37	37	37	36	35
11	Fishing Bay	0	0	0	0	0	0	25	25	25	25	24	25
12	Nantiocke & Wicomico Rivers	0	0	0	0	0	0	30	30	30	30	30	30
13	Manokin River	0	13.33	0	0	0	0	15	15	15	15	14	15
14	Patapsco River	0	5.00	0	0	0	0	21	20	21	21	-	21
15	Big & Little Annemessex Rivers	5.00	25.00	5.00	0	0	0	20	20	20	19	18	20
16	Gunpowder & Bush River Headwaters	0	0	0	0	-	0	9	8	7	9	-	9
17	Pocomoke Sound (Maryland)	0	0	0	-	0	0	22	20	21	-	20	22
18	Magothy River	16.67	0	8.33	0	-	0	12	12	12	12	-	12
19	Severn River	0	13.33	13.33	13.33	-	0	15	15	15	15	-	13
20	Patuxent River	0	0	0	0	0	0	50	47	50	50	47	49
21	Back, Middle & Gunpowder Rivers	0	0	0	0	0	0	22	22	22	22	22	22
22	Curtis & Cove Points	0	0	0	0	0	0	20	19	19	19	6	21
23	South, West & Rhode Rivers	0	0	0	0	0	0	8	10	10	8	8	8
24	Chester River	5.56	2.78	5.88	14.71	0	5.71	36	36	34	34	36	35
25	Love & Kent Points	0	0	0	0	0	0	8	8	8	8	8	8
26	Smith Island (Maryland)	0	27.27	8.33	0	0	0	17	11	12	17	17	17
	Total	1.28	5.69	2.86	1.31	0	0.48	624	615	629	610	553	628

[a] U.S. Fish and Wildlife Service Migratory Bird and Habitat Research Laboratory, 1976

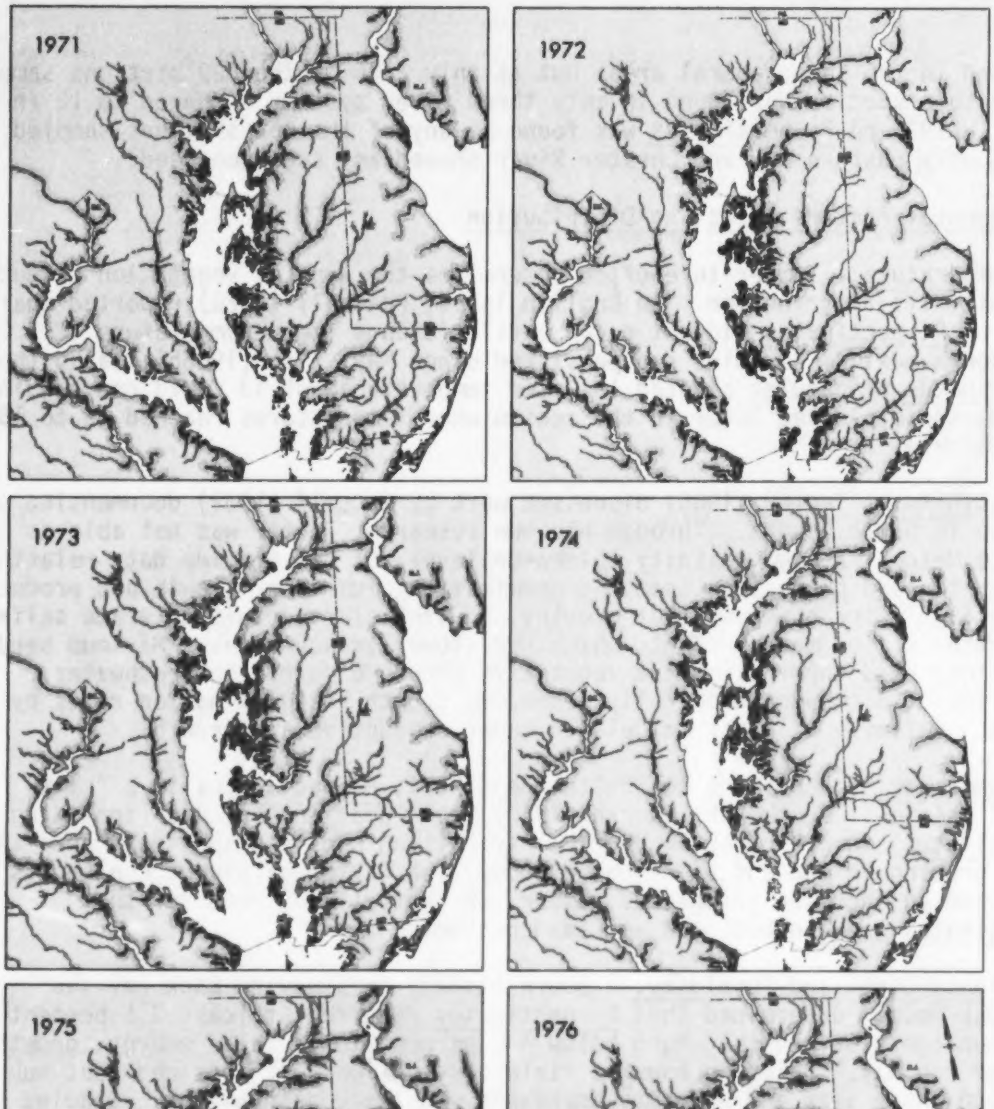

1971
1972
1973
1974
1975
1976
SCALE
NAUTICAL MILES

Figure 18. Distribution of Potamogeton pectinatus, Maryland Chesapeake Bay, 1971-1976

was found in the same general areas but at only 2.86 of the 629 stations sampled In 1974 this species was found in only three river systems compared to 12 in 1972. In 1975 no *P. pectinatus* was found at any of the 553 stations sampled. By 1976 only Eastern Bay and Chester River showed any sago pondweed.

Environmental Factors Affecting Distribution

Temperature. Temperature often determines the type of vegetation produced and limits its distribution. In English lakes, Pearsall (1920) reported that growth of *P. pectinatus* did not occur until mid-June (mean temperature 12.3 C) even though maximum sunshine had persisted since May. Yeo (1965b) stated that *P. pectinatus* propagules planted in water temperatures of 13 C did not germinate; however, those planted later in the season when temperatures reached 15 to 18 C did germinate.

Salinity. Teeter (1965) discussed work by Metcalf (1931) documenting sago pondweed in North Dakota. Through his own research, Teeter was not able to corroborate Metcalf's high salinity tolerance level but did provide data relating salinity to seed production, set and germination plus tuber growth and production. In general, Teeter observed that growing, healthy plants could tolerate salinity levels that killed mature plants which had slower growth rates. Maximum seed production, seed germination and vegetative growth occurred in freshwater. Salinities of 8 to 9 ppt generally decreased growth and germination rates by 50 percent. Salinity at 3 ppt stimulated tuber production and growth.

Substrate. According to Sculthorpe (1967), *P. pectinatus* is a "silt-loving" species. A study by Hannan (1967), concluded that plants (including *P. pectinatus*) in gravel and mud-gravel transition beds had shorter shoots than plants growing in mud and silt-pebble beds. The latter displayed continuous growth and an increase in species number. Rickett (1923) reported that *P. pectinatus* grew on both mud and sand bottoms.

Light, Depth and Turbidity. Bourn (1932), in studying Back Bay and Currituck Sound, determined that *P. pectinatus* required at least 3.5 percent total sunlight for growth. Much below 4.0 percent total solar energy, growth declined rapidly. Based on Bourn's field observations, it appeared that subterranean tuber size and quantity decreased with decreasing light. Shading of *P. pectinatus* plants produced yellowed, sparse foliage, elongated nodes and rigid unbranched stems. Stem tips and terminal leaves protruded above the water surface and wilted.

Current, Wind and Wave Action. Under conditions of strong winds and wave action, *P. pectinatus* can persist due to a lattice root structure (Hannan 1967). This may have been instrumental in sago pondweed's persistence in the Chesapeake Bay through Hurricane Agnes in June 1972. Even though the species has recently declined subsequent to the hurricane, the July 1972 sampling of the Bay by MBHRL showed the highest percent of *P. pectinatus* occurrence of the six-year survey.

Sedimentation. Sago pondweed has long, narrow, vertical leaves which do not provide a good surface for settling particles or organisms. This allows for particulate matter to settle and build up bottom sediments more conducive to invasion by other macrophytes (Butcher 1933).

Nutrient Response. Sculthorpe (1967) listed *P. pectinatus* as capable of tolerating river water polluted with sewage, other wastes and high nutrient levels that tend to limit other species. Jones and Cullimore (1973) correlated a wide range of nutrient levels in Canadian lakes with the distribution of vegetation. In all cases studied, *P. pectinatus* was a predominant species in the lakes. These studies were undertaken in late May and early June during a time when nutrient levels were critical to the development of young plants. The nutrient ranges supporting the largest populations of *P. pectinatus* included:

Nitrates	0.13 to 1.7 ppm
TKN	1.14 to 1.8 ppm
Total phosphate (dissolved)	0.6 to 1.7 ppm
Total inorganic carbon	28 to 42 ppm
Total hardness (CaCo$_3$)	346 to 405 ppm
Total organic carbon	12 to 18 ppm
Total phosphate (sediment)	0.03 to 2 ppm

Susceptibility. Heavy metals and herbicides are selectively toxic to *P. pectinatus* plants. Analysis of plant ash contents have revealed that these plants can concentrate high levels of vanadium (16,500 ppm) and lower levels of cobalt and nickel (2.8 ppm and 12 ppm respectively) (Varenko and Chuiko 1971). Plants exposed to 0.12 ppm copper in the form of copper sulfate penthydrate resulted in the development of proportionally shorter stems with yellowing of plant tissue. In the case of acute copper toxicity, the plant's outer surface became brown and internodes were shorter giving the plant a rosette appearance (Ryan and Riemer 1975).

Selectively researched herbicides, commonly used for the control of terrestrial weeds, have been reported by Yeo (1966) to be toxic to *P. pectinatus.* Experiments indicated that popularly used diquat and paraquat controlled the growth of sago pondweed. Application rates of 250 ppb of diquat or paraquat controlled plant growth for eight weeks. A concentration of 125 ppb diquat controlled plant growth in two reservoirs but was unsuccessful in two others.

Experiments by Frank et al. (1963) found that *Potamogeton* spp. were satisfactorily controlled by atrazine granules applied at 5 and 6 ppm a.i.. Simazine applied at 4.5 to 9 kg/ha controlled the growth of *Potamogeton* spp.

Newbold (1975) indicated that *P. pectinatus* was controlled in a pond for a year by concentrations of 10, 20 or 40 mg/l dichlobenil and chlorthiamid. A comparatively low concentration of 0.05 mg/l terbutryne controlled growth for one year. A concentration of 2 ppm a.i. silvex and 7 to 10 ppm a.i. sodium arsenite have been recorded to cause the cessation of *P. pectinatus* growth (Lawrence and Hollingsworth 1969).

Experiments by Devlin et al. (1972) and Devlin and Karczmarczyk (1975) on the effects of napthalam and norflurazon on sago pondweed, suggested that factors such as increased light, mineral deficiencies and growth regulators influenced the extent of herbicide damage. *P. pectinatus* plants slightly deficient in nitrogen and phosphorous took up 45 percent more napthalam than plants grown in a complete inorganic medium. Plants deficient in calcium accumulated 20 percent more napthalam, perhaps resulting from tissue breakdown due to a limited supply of calcium. Plants deficient in both magnesium and potassium took up decreased amounts of napthalam as compared to control plants. More recently, Devlin (1973, 1974) showed that sago pondweed treated with gibberilic acid,2,4-D and parachlorophenoxyacetic acid took up considerably more napthalam than untreated plants. An accelerated rate of napthalam uptake occurred with an increase in 2,4-D.

Significant data was compiled by Frank and Hodgson (1964) on the absorption of the herbicide Fenac by isolated organs of *P. pectinatus*. Exposures to a labelled Fenac solution indicated high herbicide activities in both leaves and roots.

Pathogens. Pathogenic fungi and bacteria and a brackish-water hydroid are frequently associaed with sago pondweed. The brackish-water hydroid, (*Cordylophora lacustris*) secretes a gelatinous substance and was found to result in mechanical injure (Bourn 1932). The fungus *Rhizoctonia solani* is often located between places of contact of hydroid colonies and the plant stem. Another fungal group, designated *Pythium*-3, has been determined to be the primary pathogen of *P. pectinatus* in northern Back Bay, Virginia. Evidence showed that increased salinities inhibited the growth of this fungus in lower Currituck Sound, North Carolina (Lumsden et al. 1963).

Productivity

Photosynthesis in hydrophytes is affected by the form of carbon available to plants (Steemann Nielson 1951). *Potamogeton pectinatus* has been shown to be able to assimilate HCO_3^- as an alternate carbon source for photosynthesis (Martin and Uhler 1939). In alkaline calcareous water, *P. pectinatus* often becomes encrusted with carbon deposits known collectively as marl (Sculthorpe 1967). These deposits result from the absorption and transfer of bicarbonate ions through the leaves.

Consumer Utilization

Potamogeton pectinatus is one of the more important waterfowl plant foods on this continent. It has been responsible for approximately 50 percent of the total food provided by the genus *Potamogeton* (Martin and Uhler 1939). The starchy nutlets and tubers have been reputed to be an excellent food source for ducks; rootstocks and stems are consumed to a lesser degree (Martin and Uhler 1939). Extensive beds of sago pondweed are recorded to attract Canvasbacks and Redheads to an area (Bergman 1973). Observations of Whistling Swan feeding behavior indicate that this species also consumes sago pondweed. In addition to ducks and swans, the marsh birds, geese and shorebirds are *P. pectinatus* feeders Other wildlife such as muskrat, deer and beaver have been observed feeding on *P. pectinatus* (Fassett 1960). In addition to providing a source of food, this plant provides a protective habitat for fish, oysters and benthic creatures (Fassett 1960).

Linn et al. (1972, 1975) tested the nutritional value of sago pondweed and its effects on the digestive system of sheep. Results indicated that aquatic vegetation could be an adequate forage for ruminants if economic and palatability problems could be overcome.

ZANNICHELLIA PALUSTRIS

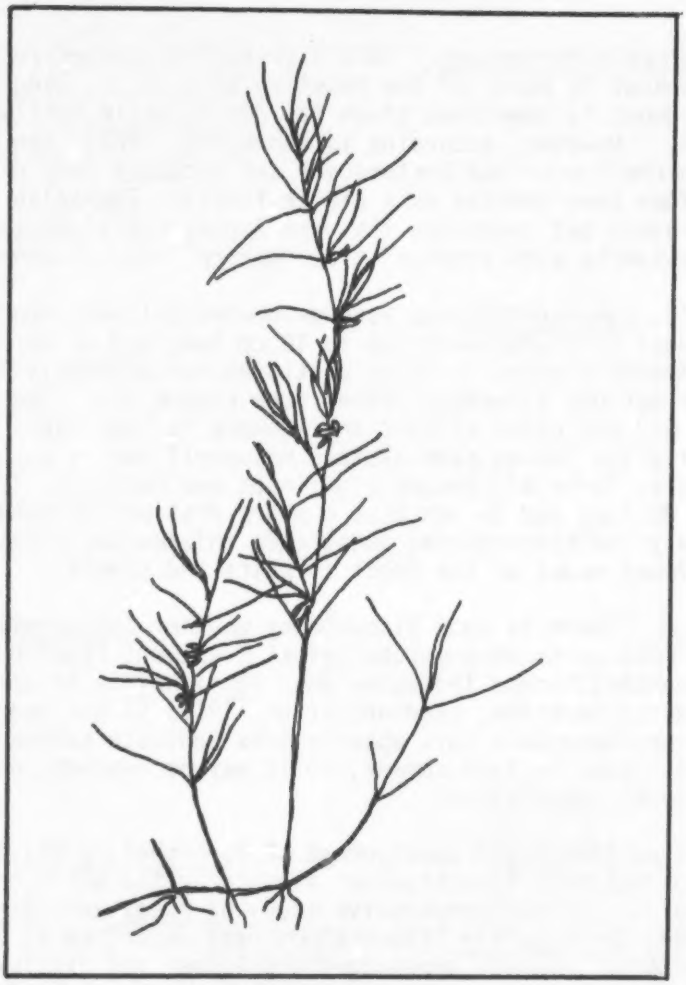
Biology

General Vegetative Morphology. This distinctive species received its generic epithet from Linnaeus in honor of the Venetian botanist G. Zannichelli. Taxonomically, this genus is sometimes given its own separate family, the Zannichelliaceae. However, according to Cronquist (1968), the Potomogetonaceae, Ruppiaceae, Zannichelliaceae and Zosteraceae are probably very closely related and they have often been treated as a single family. Takhtajan (1969) treats the family separately but indicates that the Zannichelliaceae is very near the Cymodoceaceae, a family with tropical true "marine" species (Good 1964).

Vegetatively, Zannichellia has pseudo-whorled (closely spaced numbers of subopposite leaves) filiform leaves up to 10 cm long and up to 1.0 mm wide. Its pseudo-whorled leaves enables it to be distinquished vegetatively from Ruppia which is similar but has alternate leaves (see Figure 19). The leaves have accuminate tips and are often without any stomata in some races (Burgemeister 1968). Internally the leaves have septate mesophyll and on each side of the midvein have a single, large air-lacuna (Tomlinson and Posluszny 1976). The shoots can be up to 10 dm long and do not have a sharp distinction between erect parts and the relatively undifferentiated subsurface rhizomatous portions. The roots arise from the lower nodes of the shoot in pairs and singly.

Reproduction. There is some discrepancy whether Zannichellia is a perennial or annual. The most up-to-date morphological treatment (Tomlinson and Posluszny 1976) of the Zannichelliaceae indicates that Zannichellia is ephemeral to perennial. In North Carolina, (Radford et al. 1960) it has been described as perennial. In the Chesapeake Bay, observations indicate extensive die back of the entire shoot system by late summer, so it may be reproducing mostly by seed (Stevenson, personal observation).

Recent work on the floral development of Zannichellia has indicated that Z. palustris does not have true bisexual flowers in the axils of the sub-opposite leaves as Campbells's (1897) comparative study of Najas and Zannichellia indicated. Instead, Zannichellia "flowers" are best described as "fertile nodal complexes" with three separate appendages (Posluszny and Sattler 1976). The first appendage is a membranous sheath which surrounds a second staminate and a third pistillate appendage. Both staminate and pistillate appendages branch and have renewal growth apices. Posluszny and Sattler stated that the pistillate flower develops two carpels which are eventually covered by a membranous envelope. Sometimes they found three carpels in their sample material from the Saint Lawrence River, while Fassett (1960) stated that 2 to 5 pistillate flowers have been observed. Further variation in carpel number (2 to 8) has been observed in North Carolina by Radford et al. (1964). The staminate flower always has a single stamen; however, the number of microsporangia per stamen does vary (Posluszny and Sattler 1976).



(copied from Hotchkiss 1967)

Figure 19. Horned pondweed (Zannichellia palustris)

Since the staminate flower is only a couple of millimeters away from the stigmatic surfaces of the carpels at maturity, it is possible that this species reproduces by selfing. The hyaline membranous cup around the base of the carpels may help in pollen capture. Fruit development is very rapid. In the Chesapeake Bay, seeds are found very early in the year, almost as soon as vegetative shoot growth begins. The fruit with its persistent horn-shaped style gives this species the common name of horned pondweed. Technically the fruit is classified as an achene (i.e., a small one-sided indehiscent structure with thin pericarp derived from a single carpel). The body of the achene is up to 2 to 3 mm long while the style is another 1 to 2 mm.

Seed germination of _Zannichellia_ is much more rapid than most other submerged aquatics which often germinate several years after seed set. This species has shown significant germination in the same year as the seed was set (Guppy 1897). In Lake Titicaca, Tutin (1940) found _Zannichellia_ fruiting and germinating abundantly at the same time of the year. The short life cycle of this species may help account for its rapid colonizing ability.

Distribution

Figure 20 and Table 9 show the distribution of _Zannichellia_ in the Chesapeake Bay for the 1970s based on summer survey data from MBHRL. In only two years, 1972 and 1976, was _Zannichellia_ found in abundance. Distribution was centered in the Choptank River and Eastern Bay areas. In the years 1971, 1973, 1974 and 1975, survey teams found horned pondweed at one station or less. Preliminary reports by various researchers indicate that populations may have returned to higher levels in 1977. Confirmation from MBHRL is not available at this time. This distribution pattern is among the more erratic of all the Chesapeake Bay submerged aquatic species. To some extent, this may be an artifact of summer surveying. _Zannichellia_ populations seem to decline rapidly in June and early July, so that late July and August surveys might not be adequate to monitor this species. However, _Zannichellia_ does appear to be an ephemeral pioneer species which does show considerable yearly fluctuations in occurrence off Horn Point in the Choptank River (Stevenson, personal communication). Fenwick (personal communication) has studied _Zannichellia_ abundance in Eastern Bay and has suggested that _Zannichellia_ can be easily missed in beds of mixed species. After its early growing period, _Zannichellia_ beds are often over run by other morphologically similar species, (i.e. _Ruppia_). Dense _Ruppia_ beds tend to hide the shorter _Zannichellia_ plants.

Historically in the Chesapeake Bay, this species seems to be a relative "late comer" in the herbarium record. The earliest reported specimen from a Bay area herbarium dates from 1931 in the Cove Point area on the Western Shore. After that, there have been relatively few records until the 1960s. Most other submerged aquatic species are documented in herbaria before the 1900s in the Chesapeake Bay. This may indicate that _Zannichellia_ has been established relatively recently or at least was in relatively low frequencies in the Chesapeake in the past. However, paleopollen studies of Bay sediments would be necessary to confirm or reject this hypothesis.

Zannichellia palustris is remarkably cosmopolitan. Clapham et al. (1952) reported that it was a native of the British Isles and is locally common in

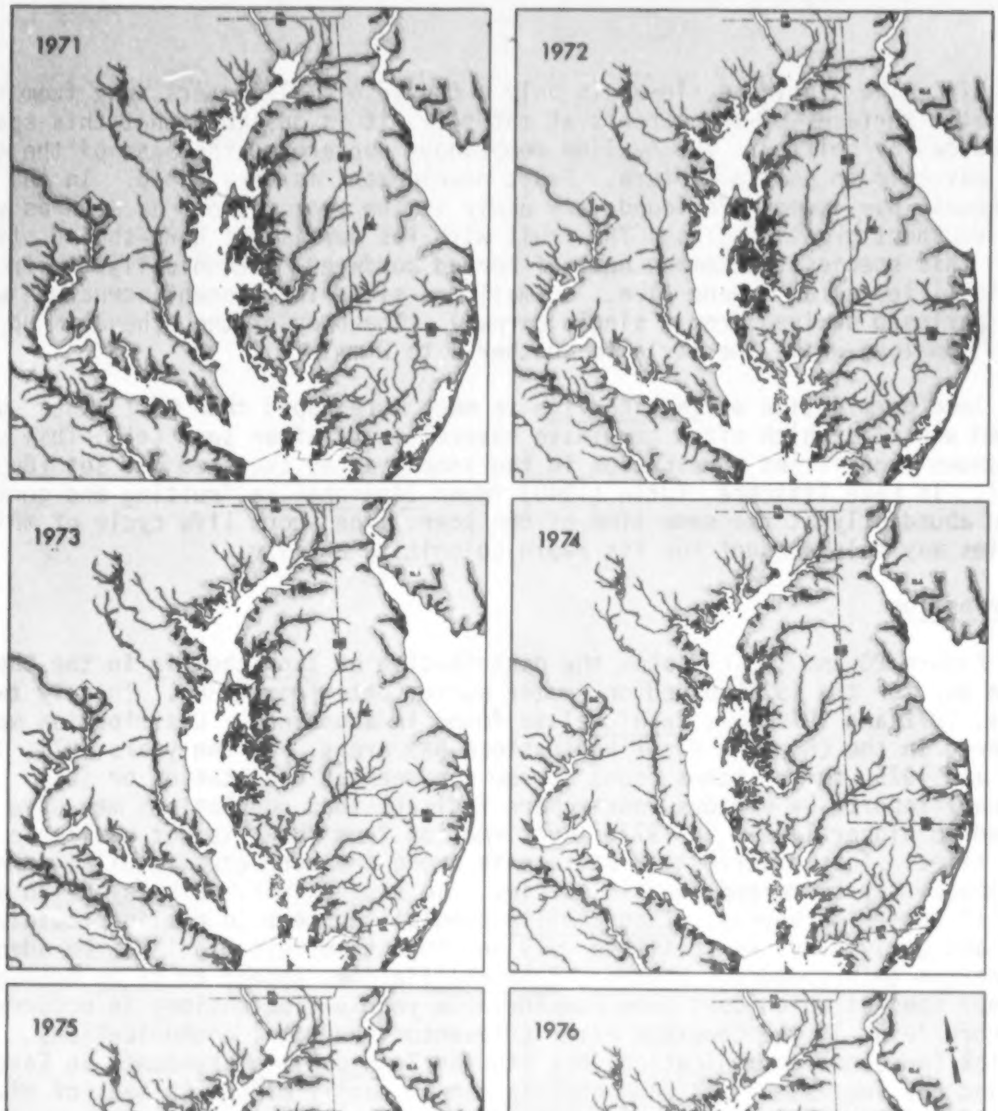

1971
1972
1973
1974
1975
1976
SCALE
NAUTICAL MILES

Figure 20. Distribution of *Zannichellia palustris*, Maryland Chesapeake Bay, 1971-1976

Table 9. Percent of sampling stations showing occurrence of *Zannichellia palustris*, Maryland Chesapeake Bay, 1971-1976[a]

Area Code	River System	1971	1972	1973	1974	1975	1976	Number of stations					
								71	72	73	74	75	76
1	Elk & Bohemia Rivers	0	0	0	0	0	0	15	16	16	16	16	16
2	Sassafras River	0	0	0	0	0	0	10	10	10	10	10	10
3	Howell & Swann Points	0	0	0	0	0	0	12	6	12	12	12	12
4	Eastern Bay	0	13.95	0	0	0	20.00	47	43	47	47	46	45
5	Choptank River	0	17.24	0	0	0	14.29	60	58	57	58	57	56
6	Little Choptank River	0	5.26	0	0	0	0	19	19	19	19	19	19
7	James Island & Honga River	0	0	0	0	0	2.94	34	34	34	34	34	34
8	Honga River	0	0	0	0	0	3.45	30	30	30	30	29	29
9	Bloodsworth Is.	0	0	0	0	0	0	40	44	46	43	43	45
10	Susquehanna Flats	0	0	0	0	0	0	27	37	37	37	36	35
11	Fishing Bay	0	0	0	0	0	0	25	25	25	25	24	25
12	Nanticoke & Wicomico Rivers	0	0	0	0	0	0	30	30	30	30	30	30
13	Manokin River	0	0	0	0	0	0	15	15	15	15	14	15
14	Patapsco River	0	0	0	0	0	0	21	20	21	21	0	21
15	Big & Little Annemessex Rivers	0	0	0	0	0	0	20	20	20	19	18	20
16	Gunpowder & Bush River Headwaters	0	0	0	0	-	0	9	8	7	9	-	9
17	Pocomoke Sound (Maryland)	0	0	0	-	0	0	22	20	21	-	20	22
18	Magothy River	0	0	0	0	-	0	12	12	12	12	-	12
19	Severn River	0	0	0	6.67	-	7.69	15	15	15	15	-	13
20	Patuxent River	0	0	0	0	0	0	50	47	50	50	47	49
21	Back, Middle & Gunpowder Rivers	0	0	0	0	0	0	22	22	22	22	22	22
22	Curtis & Cover Points	0	0	0	0	0	0	20	19	19	19	6	21
23	South, West & Rhode Rivers	0	0	0	0	0	0	8	10	10	8	8	8
24	Chester River	0	0	2.94	0	0	5.71	36	36	34	34	36	35
25	Love & Kent Points	0	0	0	0	0	0	8	8	8	8	8	8
26	Smith Island (Maryland)	0	0	0	0	0	0	17	11	12	17	17	17
	Total	0	2.76	0.16	0.16	0	3.50	624	615	629	610	553	628

[a] U.S. Fish and Wildlife Service Migratory Bird and Habitat Research Laboratory, 1976.

rivers, streams, ditches and brackish pools. It is also widespread throughout Europe where Reese (1963) documented several chromosome races. The species is also well distributed in the Old World.

In North America, Porsild (1932) reported Zannichellia palustris within the Artic Circle on the north coast of the Seward Penninsula in the Buckland River delta region. Although this species has been documented in every state in the continental United States (Fassett 1960), it is not a commonly occurring submerged aquatic. Deane (1910) reported that Zannichellia could be found only occasionally in brackish marshes along the New England coast, but it rarely occurred inland. South of the Chesapeake Bay, this species can be found in abundance in the Currituck and Pamlico Sound areas in North Carolina. However, in South Carolina and the southern Coastal Plain counties of North Carolina, Zannichellia is relatively rare.

In South America, this species has been found in Lake Titicaca located at 3,815 m in the Andes Mountains (Hutchinson 1975). The widespread occurrence of this species even at such remote locations suggests excellent dispersal mechanisms as well as a wide range of adaptability to varying climates and altitudes.

Environmental Factors Affecting Distribution

Temperature. Comparatively little information is available from the literature on how physical environment and water quality parameters affect this species. There are indications in the Chesapeake Bay that this species may be sensitive to high water temperatures. As summer approaches and water temperatures increase above 30 C, Zannichellia populations often begin to decline rapidly. However, it is unclear whether this is a strict temperature effect or whether other variables are involved. Fenwick (personal communication) found healthy Zannichellia growing in Eastern Neck (Eastern Bay area) tidal pools where the temperature often exceeded 38 C. Porsild (1932) indicated that this species was found in hot springs in Kamchatka (USSR) where water temperatures varied from 25 to 30 C. At the lower end of the temperature range is Lake Titicaca where Tutin (1940) reported Zannichellia populations existed in a temperature range from 10.5 to 14.8 C.

Salinity. Horned pondweed is able to persist in fresh water, but there seems to be a definite preference of this species for brackish areas or calcareous pools (Radford et al. 1964) and hard water situations (Fassett 1960). In the Chesapeake Bay this species has been found as far south as the mouth of the Honga River. Salinities seldom go beyond 20 ppt in this area so this may be the maximum limit of salinity tolerance for Zannichellia in the Chesapeake Bay.

Sediments. Zannichellia has a tendency to grow in clay to sandy sediments in shallower water than other submerged aquatics in the Chesapeake Bay. This is also the case in Lake Titicaca where Tutin found Zannichellia in the shallowest situations whereas Potamogeton, Myriophyllum and Elodea dominated the deeper waters. This suggests that Zannichellia may need higher light intensities than other submerged aquatics for optimal growth. Correll et al. (1977) subjected Zannichellia to 80 to 140 $\mu E/m^2/sec$ at the water surface (approximately

4 to 7 percent of the maximum noon summer sunlight)and obtained good growth. Little else is available concerning the light requirements of this species.

Current, Wind and Wave Action. In the Chesapeake, *Zannichellia* grows in shallow areas where up to one knot currents develop. Sculthorpe (1967) mentioned that the adventitious roots of this species are spiral, tendril-like and aid in anchorage. The rhizomes of this species are very slender and would otherwise be easily dislodged in strong currents without the adventitious root system. However, this adaptation is apparently not effective where wave action is significant, since this species is never found on high energy shorelines.

Zannichellia was one of the few submerged aquatics that did well in the Chesapeake Bay after Hurricane *AGNES* when much sediment was deposited on the Bay bottom. The adaptive strategy of this species in regard to sedimentation seems to be toward rapid recolonization, rather than growing up through sediments which are being rapidly deposited. No information is available on how well this species does in regard to trapping sediments and reducing turbidity. However, its filiform leaves and slender rhizomes are probably not as effective in this regard as *Myriophyllum* and other species which grow in thick masses.

In terms of water quality responses, *Zannichellia* may be associated with increased nutrient loading. In reviewing some of the work on central European Lakes, Hutchinson (1975) indicated that *Zannichellia* was an important species in eutrophic waters. In mesotrophic locations, *Najas* displaced *Zannichellia* as a codominant with narrow leaved *Potamogeton* species. *Zannichellia* has been found in recent surveys of Lake Mendota. This lake has been undergoing rapid eutrophication since the early 1900s. Hutchinson (1975) cited early surveys of Lake Mendota in 1912, 1922 and 1940 as having found no *Zannichellia*. Recent work by Lind and Cottam (1969) showed that *Zannichellia* appeared in the lake in the shallowest emersed zone while *Myriophyllum exalbescens* formed monospecific stands in deeper waters. Since *Zannichellia*, as mentioned previously, has only been widely established in the Chesapeake Bay since the 1960s, it is possible that its spread may be an indicator of increased eutrophication of the Bay. However, *Zannichellia* is not found in all eutrophic waters of the Chesapeake Bay.

Productivity

The only productivity data available for *Zannichellia* comes from *in situ* measurements in the Rhode River, a subestuary of the Chesapeake Bay. Correll et al. (1977) measured oxygen metabolism of plants placed in glass bottles at depths of 20 and 40 cm below the surface. At 20 cm net oxygen production was calculated to be 462 mg O_2/g dry weight tissue while at 40 cm it was 298 mg O_2/g dry weight tissue. Net photosynthesis was 26.0 mg O_2/hr/g dry weight at 20 cm and 22.0 mg O_2hr/g dry weight at 40 cm. Respiration measured in dark bottles was 6.3 mg O_2/hr/g dry weight at 20 cm and 7.1 mg O_2/hr/g dry weight at 40 cm. Unfortunately, there is presently little comparative data for other submerged aquatics using similar techniques.

Consumer Utilization

Fassett (1960) reported that fruits and sometimes foliage of *Zannichellia* are good for waterfowl in brackish pools. Furthermore, it was rated by

Needham (1938) as a fair food producer for trout. However, *Ruppia*, *Vallisneria* and *Potamogeton* spp. are usually considered more important in nourishing fish and wildlife species.

Economic Uses and Problems

This species has not reached the epidemic proportion in terms of population size as did *Myriophyllum*. Therefore, there are no particular problems in the Chesapeake Bay associated with its occurrence. Since it is a pioneer species, it may be helpful in revegetating areas where it is desirable to have submerged aquatics. Little use has been made of *Zannichellia* for this purpose up to the present time.

VALLISNERIA AMERICANA

Biology

General Vegetative Morphology. Often referred to as wild celery, tape grass or eel-grass, *Vallisneria americana* belongs to the Hydrocharitaceae family along with *Elodea*. *V. americana*, *V. asiatica* and *V. aethiopica* are thought by some botanists to be geographic races of *V. spiralis* (Sculthorpe 1967). Fassett (1960) used the name *V. americana* and listed *V. spiralis* as a synonym referenced with Gray's Manual and Britton and Brown. The name *V. americana* will be used for the purpose of this document.

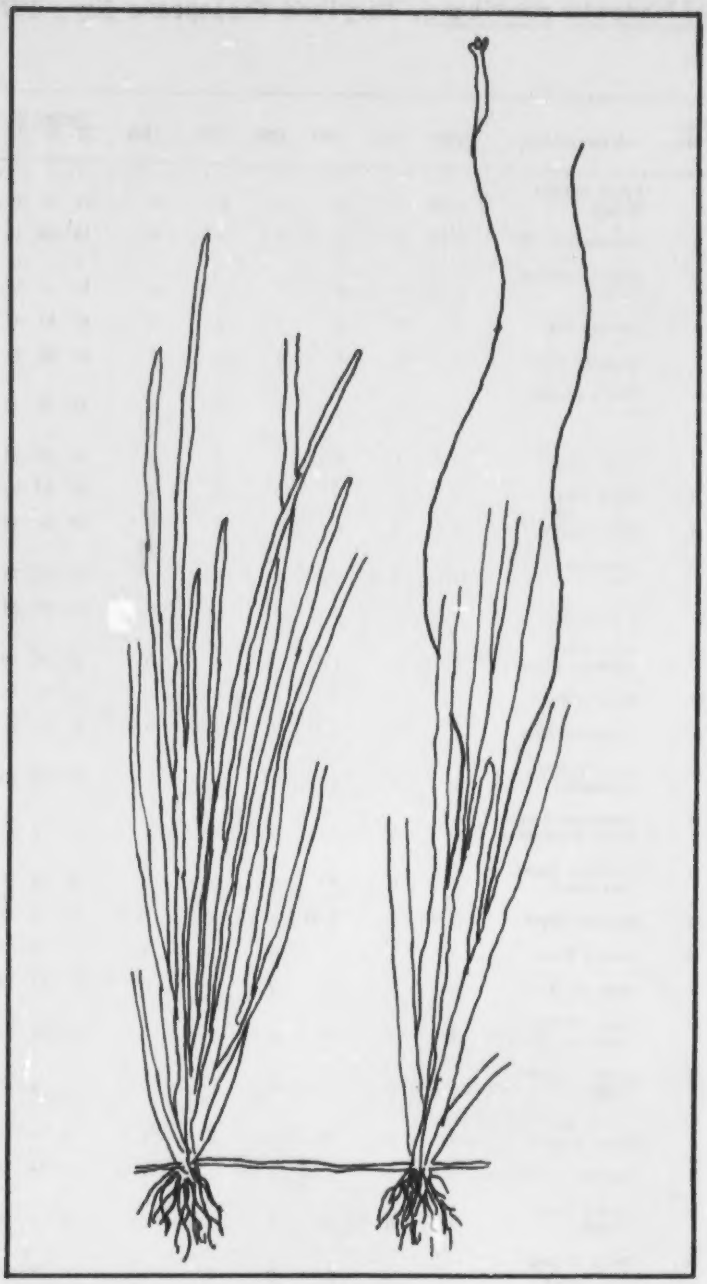
Vallisneria americana is a monocotolydon, with ribbon or strap-shaped translucent leaves, 3 to 10 mm wide and arranged in basal clusters at the ends of creeping stems (Schuette and Alder 1927; Sculthorpe 1967; Radford et al. 1964; Needham and Lloyd 1930) (see Figure 21). Leaves may attain up to 1.8 m in length with the height of the plant more or less limtied by water depth (Schuette and Alder 1927).

Reproduction. Vegetative propogation occurs primarily through the vigorous growth of stolons or runners and through tubers (Sculthorpe 1967; Lamoureux 1957). Sexual reproduction is more complex. Pistillate flowers are borne on peduncles that elongate at anthesis until they reach the water surface. Staminate flowers, by comparison, are free-floating after having broken away from the plant base at anthesis. Once fertilization has occurred at the water/air interface, the pistillate peduncles coil downwards and submerge the many seeded, berry-like fruit (Sculthorpe 1967; Muenscher 1936).

Distribution

Wild celery is considered to be a fresh water macrophyte occurring in the tidal streams of the Atlantic Coastal Plain (Martin and Uhler 1939).

The summer sampling conducted by MBHRL has documented *V. americana* in the Chesapeake Bay from 1971 through 1976. Though the data is sparse (see Table 10 and Figure 22) species occurrence shows a decline over the six year period. In 1971, wild celery was well represented in the Susquehanna Flats. By 1973, those same stations were bare of wild celery and subsequent sampling has not shown any



(copied from Hotchkiss 1967)

Figure 21. Wildcelery (Vallisneria americana)

Table 10. Percent of sampling stations showing occurrence of *Vallisneria americana*, Maryland Chesapeake Bay, 1971-1976[a]

Area Code	River system	1971	1972	1973	1974	1975	1976	Number of stations					
								71	72	73	74	75	76
1	Elk & Bohemia Rivers	6.67	0	0	0	0	0	15	16	16	16	16	16
2	Sassafras River	20.00	0	0	0	0	0	10	10	10	10	10	10
3	Howell & Swann Points	8.33	0	0	0	0	0	12	6	12	12	12	12
4	Eastern Bay	0	0	0	0	0	0	47	43	47	47	46	45
5	Choptank River	0	0	0	0	0	0	60	58	57	58	57	56
6	Little Choptank River	0	0	0	0	0	0	19	19	19	19	19	19
7	James Island & Honga River	0	0	0	0	0	0	34	34	34	34	34	34
8	Honga River	0	0	0	0	0	0	30	30	30	30	29	29
9	Bloodsworth Is.	0	0	0	0	0	0	40	44	46	43	43	45
10	Susquehanna Flats	37.04	2.70	0	0	0	0	27	37	37	37	36	35
11	Fishing Bay	0	0	0	0	0	0	25	25	25	25	24	25
12	Nanticoke & Wicomico Rivers	0	0	0	0	0	0	30	30	30	30	30	30
13	Manokin River	0	0	0	0	0	0	15	15	15	15	14	15
14	Patapsco River	0	0	4.76	9.52	-	9.52	21	20	21	21	-	21
15	Big & Little Annemessex Rivers	0	0	0	0	0	0	20	20	20	19	18	20
16	Gunpowder & Bush River Headwaters	0	0	0	0	-	0	9	8	7	9	-	9
17	Pocomoke Sound (Maryland)	0	0	0	0	0	0	22	20	21	0	20	22
18	Magothy River	16.67	0	8.33	0	-	8.33	12	12	12	12	-	12
19	Severn River	0	0	0	0	-	0	15	15	15	15	-	13
20	Patuxent River	0	0	0	0	0	0	50	47	50	50	47	49
21	Back, Middle & Gunpowder Rivers	0	0	0	4.55	9.09	0	22	22	22	22	22	22
22	Curtis & Cove Points	0	0	0	0	0	0	20	19	19	19	6	21
23	South, West & Rhode Rivers	0	0	0	0	0	0	8	10	10	8	8	8
24	Chester River	2.78	0	2.94	5.88	8.33	2.86	36	36	34	34	36	35
25	Love & Kent Points	0	0	0	0	0	0	8	8	8	8	8	8
26	Smith Island (Maryland)	0	0	0	0	0	0	17	11	12	17	17	17
	Total	2.72	0.16	0.48	0.82	0.90	0.64	624	615	629	610	553	628

[a] U.S. Fish and Wildlife Service Migratory Bird and Habitat Research Laboratory, 1976

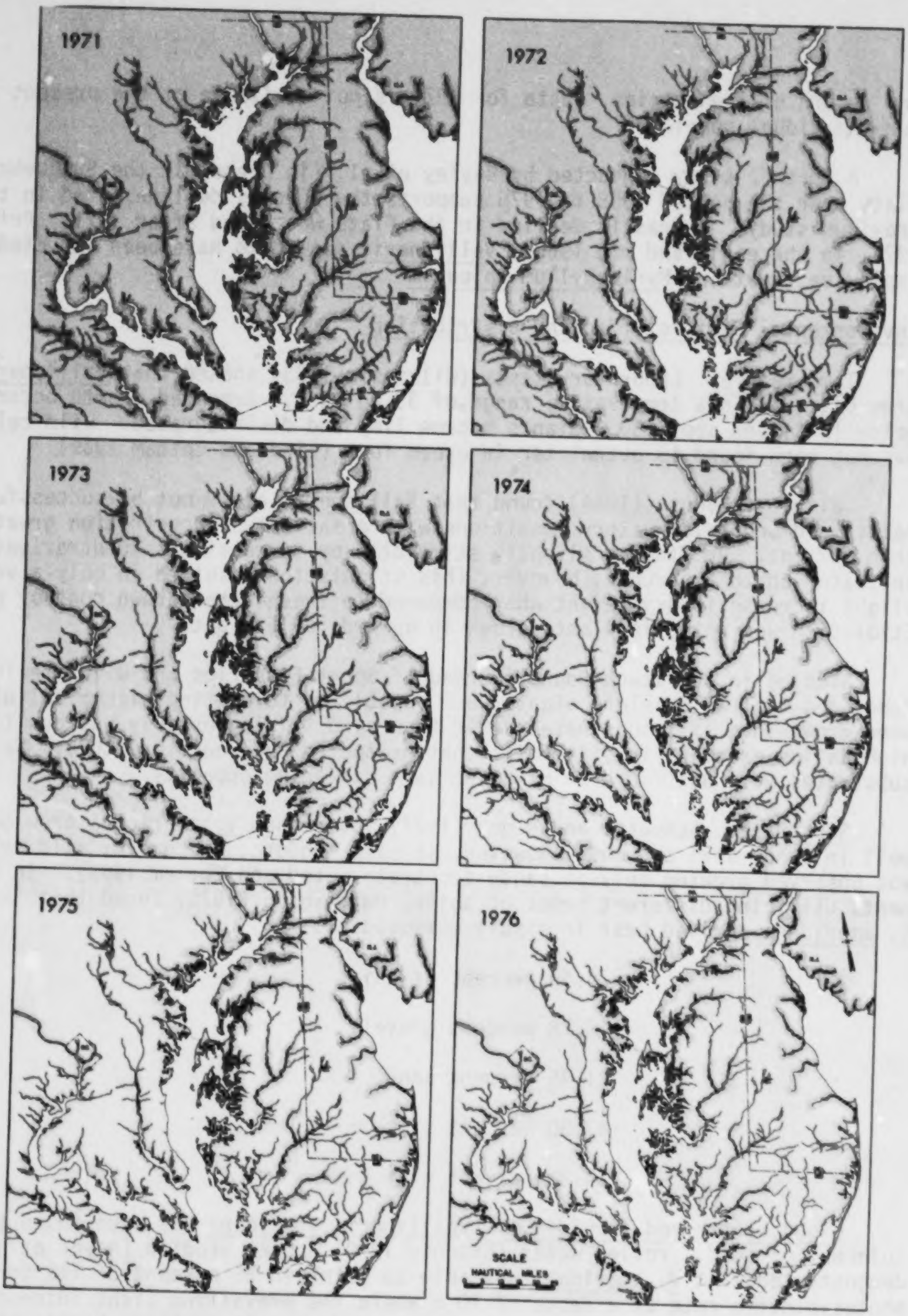

1971
1972
1973
1974
1975
1976
SCALE
NAUTICAL MILES

Figure 22. Distribution of Vallisneria americana, Maryland Chesapeake Bay, 1971-1976

any return of the species. Data for 1977 is not available at the present time for individual species.

A further study conducted by Bayley et al. (in press) in the Susquehanna Flats over the period 1958 to 1975 supports the species decline noted in the previous study. A drastic decline in the Flats was noted after 1971. Prior to 1971, in the early and mid 1960s, *Vallisneria* seemed to have been impacted by the explosive growth of *Myriophyllum spicatum*.

Environmental Factors Effecting Distribution

Temperature. Laboratory tests (Wilkinson 1963) showed that *Vallisneria* grew best within a temperature range of 33 to 36 C. Arrested growth occurred below 19 C; and above 50 C plants became limp and disintegrated. Wild celery has not been found to overwinter in green form (Lind and Cottam 1969).

Salinity. Bourn (1934) found that *Vallisneria* could not be successfully maintained under laboratory conditions with a salinity concentration greater than 4.2 ppt. Growth was slightly stimulated by increasing concentrations of saltwater up to 8 percent; however, this stimulation resulted in only a very slight increase in dry weight when compared to fresh-water grown control plants. Etiolation was noted in plants grown in upwards of 6.6 ppt.

Studies in Currituck Sound (Bureau of Sport Fisheries and Wildlife 1965) concluded that wild celery plants were capable of tolerating higher salinities when grown in a silt substrate rather than sand. This possibly was due to the high cation exchange in silt soils that protected the root structure. Sand substrates were not found to have the same buffering capacity.

Substrate. Schuette and Alder (1927) found that *V. americana* grew equally well in sandy soil and mud. However, in Lake Mendota, Wisconsin, wild celery was observed growing only on sandy substrate (Lind and Cottam 1969). In experiments utilizing different types of soils, Hutchinson (1975) found that *V. americana* thrived best in a soil composed of:

6.50 percent organics

8.78 percent gravel

21.46 percent sand

47.90 percent silt

14.26 percent clay

Light, Depth and Turbidity. *Vallisneria americana* has been found to be tolerant of muddy, roiled water (Steenis 1970). Lake studies (Meyer et al. 1943) demonstrated that *V. americana* was able to maintain 25 percent of its surface photosynthetic rate at a depth of 10 m where the prevailing light intensity was 0.5 percent of surface light on the clearest days. In northern Currituck Sound, North Carolina, during the summer and fall, 1977, *Vallisneria americana* was invariably found in shallow water (0.5 to 1.0 m) compared to *Myriophyllum spicatum* which was dominant in the deeper areas (1.0 to 2.0 m). This suggests that Vallisneria may require higher light intensities for optimal growth than other SAV species, specifically milfoil (Confer, personal observation).

Current Velocity, Wind and Wave Action. Extensive damage to Vallisneria can occur from severe natural wave turbulence and from the action of motorboat propellers (Lamoureux 1957).

Nutrient Responses. In assessing the role of Vallisneria in nutrient cycles, wild celery was found to remove more silica, phosphorus, iron, aluminum, manganese, lime, potassium and sodium than Potamogeton spp. but less sulphur, nitrogen and magnesium (Schuette and Alder 1927).

Susceptibility. Table 11 lists various chemical agents that have been used experimentally to eradicate Vallisneria americana.

Table 11. Effects of selected herbicides on Vallisneria americana[a]

Herbicide	Application rate	Central
Silvex	0.5 to 2 ppm 5 to 30 ppm 2 ppm	None 70 to 100% Satisfactory
Diquat dibromide	0.25 ppm	30 to 90%
Diquat dichloride	0.25 to 0.5 ppm	30 to 100%
Paraquat dichloride	0.25 to 0.5 ppm	67 to 93%
2,4-D IOE	16.5 to 27.5 kg/ha granules	Unsatisfactory
2,4-D D, E	44 kg/ha granules	None
Acrolein	2.55 ppm	Killed
Endothall, DSS	2 to 3 ppm	Seasonal

[a] Lawrence and Hollingsworth 1969

Consumer Utilization

All parts of the plant structure of V. americana are consumed by fish, ducks, coots, geese, grebes, swans, waders, shore and game birds. Wild celery further serves as a shade, shelter and spawning medium for fish. (Sculthorpe 1967). Abundant aquatic insect life has also been observed on and about wild celery plants (Fassett 1960; Lamoureux 1957).

An infusion prepared from Vallisneria leaves has historically served as an invigorating tonic (Sculthorpe 1967).

Economic Uses and Problems

Impoundments. Use has been made of Vallisneria americana for waterfowl food in waterfowl management in the southeast (Ball 1965). However, due to natural plant succession and almost certain invasion by pest plants in fresh or low saline areas, beneficial results were found to be temporary unless strenuous management practices were employed.

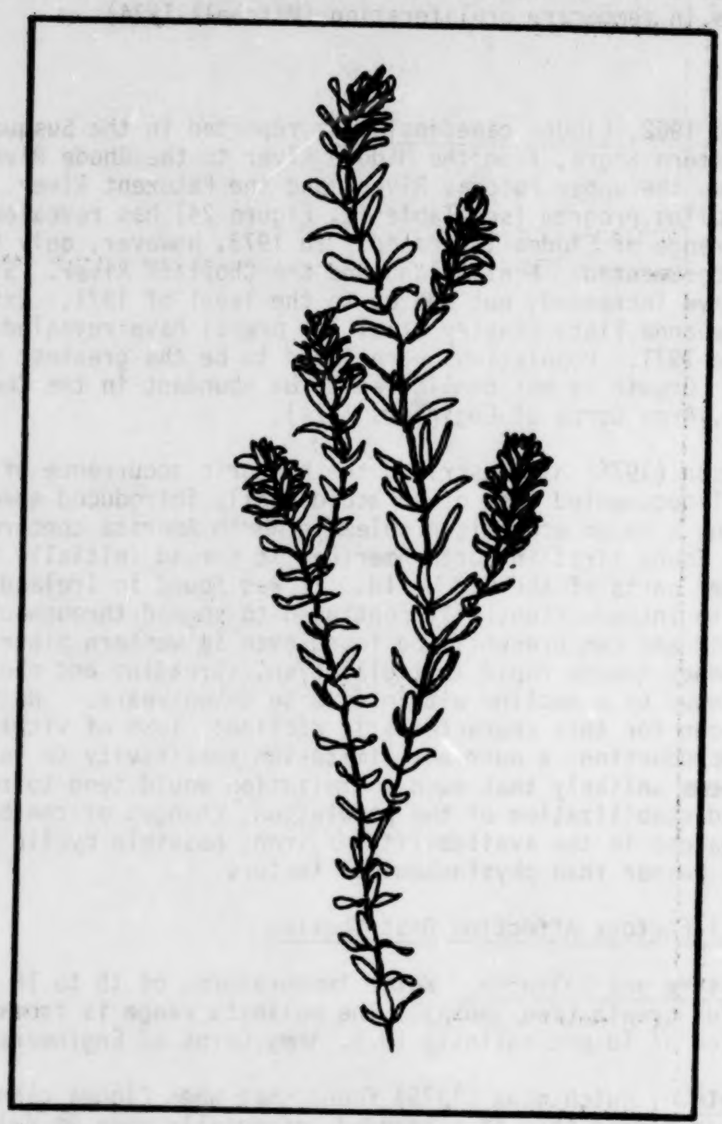
ELODEA CANADENSIS

Biology

General Vegetative Morphology. Elodea canadensis is a member of the Hydrocharitaceae family and is commonly referred to as waterweed (U.S. Army Corps of Engineers 1974). Endemic to North America and naturalized to many industrialized nations of Europe and the southern hemisphere, this species has long been recognized as an aquatic weed with great ability to colonize new environments and nutrient enriched waters (Adams et al. 1971). It is also known as both Anacharis and Elodea planchonii. The later name results from the taxonomic classification of a rare hermaphroditic condition of Elodea canadensis. The term Anacharis is an acceptable name which includes both dioecious and hermaphrodite species. Elodea canadensis, however, applies only to the dioecious species which is characteristic of those normal specimens found in North America (Weatherby 1932).

Mason (1969) characterized Elodea as having slender stems and leaves in threes with the lower ones opposite, ovate-oblong, acute, whorled, crowded and overlapping near the tips (see Figure 23). The staminate flowers are solitary, long-remaining and attached by a long peduncle, while the pistillate flowers are solitary and sessile. The fruit is linear or lanceolate-linear. The plants are submersed with branching stems forming large masses near the bottom (Fassett 1960).

Fragments of Elodea have been known to survive lengthy periods of exposure (Hutchinson 1975). Introduction to formerly uncolonized areas has commonly lead to massive proliferation. Evidence suggests that ordinarily large populations that develop after a region has been invaded are temporary and may be expected to decline naturally without apparent reason. Portions that break off from the plant do not usually die, but regenerate to form new plants. Efficient production of a large number of vegetative propagules has allowed almost complete independence from sexual reproduction (Mitchell 1974). These segments, including stem apices and small branches, are usually specialized asexual propagules that can overwinter (Yeo 1965b).



(copied from Hotchkiss 1967)

Figure 23. Common elodea (Elodea canadensis)

Reproduction. Elodea is rarely seen to reproduce sexually and the dominant plants found are female. It is believed that Elodea in Europe consists of only female plants. Therefore, reproduction is predominantly asexual resulting from the fragmentation of parent plants. Detached pieces form adventitious roots and often become the "face" of new infestations. Population declines are believed to result from a decline in viability but introduction to a new regions often results in temporary proliferation (Mitchell 1974).

Distribution

Prior to 1962, Elodea canadensis was reported in the Susquehanna Flats; the upper Eastern Shore; from the Middle River to the Rhode River on the Western Shore; the upper Potomac River; and the Patuxent River. The MBHRL Maryland sampling program (see Table 12, Figure 24) has revealed that from 1971 to 1972 the range of Elodea increased. In 1973, however, only two reports of Elodea were documented: Kent Island and the Choptank River. Since 1974, Elodea appears to have increased, but not up to the level of 1971. Extensive studies of the Susquehanna Flats (Bayley et al. in press) have revealed a population decline since 1971. Populations were found to be the greatest in the early and late 1960s. Growth is not considered to be abundant in the Chesapeake Bay as a whole (U.S. Army Corps of Engineers 1974).

Hutchinson (1975) has described the historic occurrence of Elodea. The earliest well-documented case of an accidentally introduced species of an aquatic plant becoming a major economic problem in North America concerns Elodea canadensis. Found first in North America, it spread initially to Europe and later to other parts of the Old World. It was found in Ireland in 1836 through three separate introductions. It continued to spread throughout England, entered France in 1850 and can presently be found even in western Siberia. There seems to be a tendency toward rapid multiplication, spreading and choking of water systems followed by a decline within five to seven years. Hutchinson suggests several reasons for this characteristic decline: loss of vitality owing to lack of sexual reproduction; a nutrient limitation sensitivity to ferrous iron, although it seems unlikely that such a limitation would tend to result in variable reduction and stabilization of the population; changes of the bottom sediments involving changes in the availability of iron; possible cyclic nature; and perhaps, biotic rather than physiochemical factors.

Environmental Factors Affecting Distribution

Temperature and Salinity. Water temperatures of 15 to 18 C are necessary for successful growth (Yeo 1965b). The salinity range is from fresh water to brackish water of 10 ppt salinity (U.S. Army Corps of Engineers 1974).

Substrate. Hutchinson (1975) found that when Elodea plants were rooted they grew much better than if suspended, especially when in calcareous waters. Yeo (1965b) determined that Elodea preferred a soil to sand substrate, developing roots originating at the nodes. Elodea has been found to be an important member of the deep water community in Poland and England. Rooted in stream sediment, Elodea did much better than many other plants (Hutchinson 1975).

Table 12. Percent of sampling stations showing occurrence of *Elodea canadensis*, Maryland Chesapeake Bay, 1971-1976[a]

Area Code	River System	1971	1972	1973	1974	1975	1976	Number of stations					
								71	72	73	74	75	76
1	Elk & Bohemia Rivers	0	0	0	0	0	0	15	16	16	16	16	16
2	Sassafras River	0	0	0	0	0	0	10	10	10	10	10	10
3	Howell & Swan Points	0	0	0	0	0	0	12	6	12	12	12	12
4	Eastern Bay	0	4.65	2.13	4.26	0	6.67	47	43	47	47	46	45
5	Choptank River	6.67	6.90	1.75	1.72	0	1.79	60	58	57	58	57	56
6	Little Choptank River	0	0	0	0	0	0	19	19	19	19	19	19
7	James Island & Honga River	0	0	0	0	0	0	34	34	34	34	34	34
8	Honga River	0	0	0	0	0	0	30	30	30	30	29	29
9	Bloodsworth Is.	0	0	0	0	0	0	40	44	46	43	43	45
10	Susquehanna Flats	33.33	0	0	0	0	0	27	37	37	37	36	35
11	Fishing Bay	0	0	0	0	0	0	25	25	25	25	24	25
12	Nanticoke & Wicomico Rivers	0	0	0	0	0	0	30	30	30	30	30	30
13	Manokin River	0	0	0	0	0	0	15	15	15	15	14	15
14	Patapsco River	0	0	0	0	-	0	21	20	21	20	-	21
15	Big & Little Annemessex Rivers	0	0	0	0	0	0	20	20	20	19	18	20
16	Gunpowder & Bush River Headwaters	0	0	0	0	-	0	9	8	7	9	-	9
17	Pocomoke Sound (Maryland)	0	0	0	-	0	0	22	20	21	-	20	22
18	Magothy River	0	0	0	0	-	0	12	12	12	12	-	12
19	Severn River	0	0	0	0	-	0	15	15	15	15	-	13
20	Patuxent River	0	0	0	0	0	0	50	47	50	50	47	49
21	Back, Middle & Gunpowder River	0	4.55	0	0	0	0	22	22	22	22	22	22
22	Curtis & Cover Points	0	0	0	0	0	0	20	19	19	19	6	21
23	South, West & Rhode Rivers	0	0	0	0	0	0	8	10	10	8	8	8
24	Chester River	16.67	5.56	0	0	5.56	2.86	36	36	34	34	36	35
25	Love & Kent Points	0	0	0	0	0	0	8	8	8	8	8	8
26	Smith Island (Maryland)	0	0	0	0	0	0	17	11	12	17	17	17
	Total	3.04	1.46	0.32	0.49	0.36	0.80	624	615	629	610	553	628

[a] U.S. Fish and Wildlife Service Migratory Bird and Habitat Research Laboratory, 1976

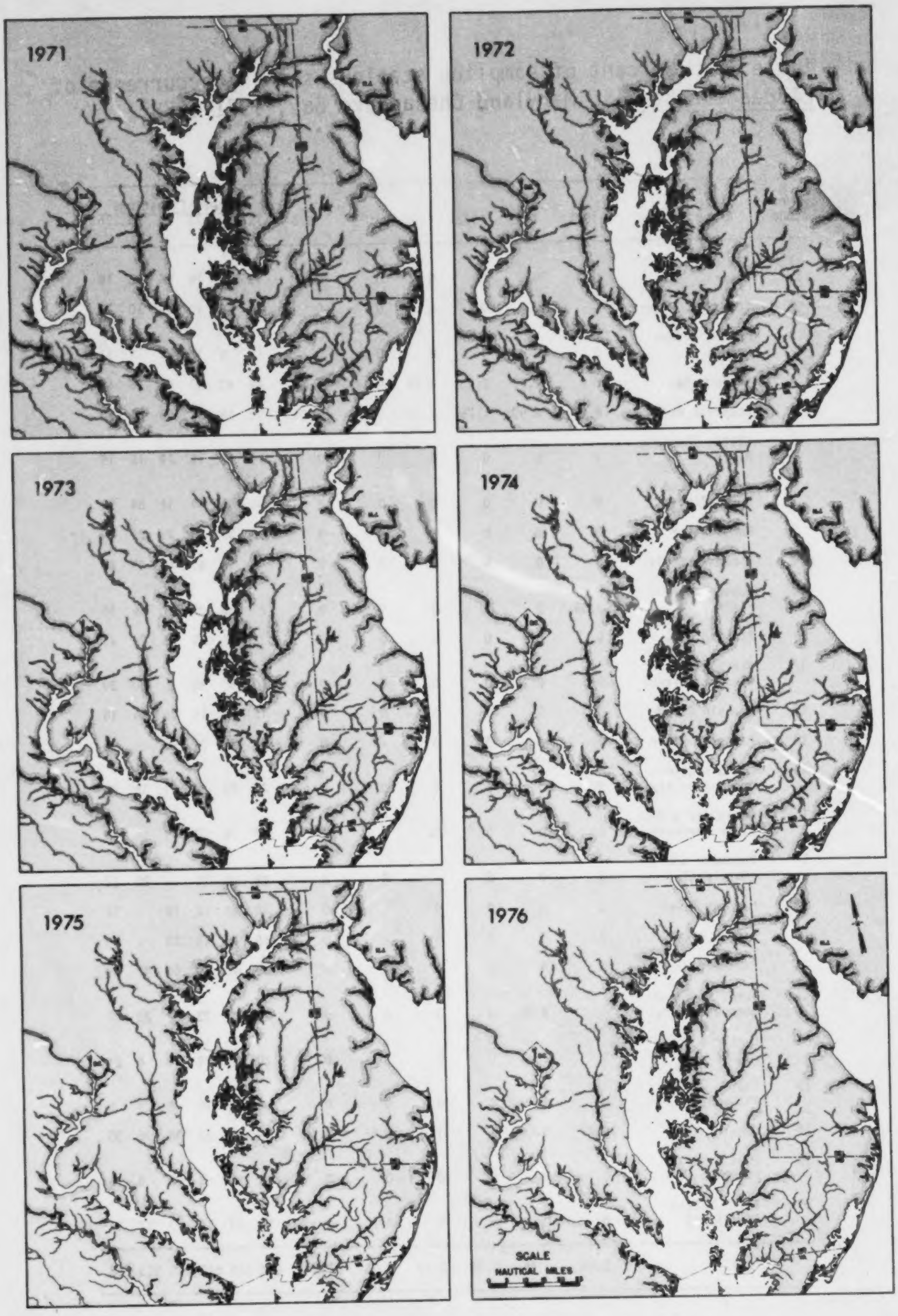

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Figure 24. Distribution of Elodea canadensis, Maryland Chesapeake Bay, 1971-1976

Light, Depth and Salinity. The maximum frequency of *Elodea* occurrence has been found at a depth of about 3.0 m and a lower limit of 7.5 m (Hutchinson 1975). *Elodea* does not appear to be sensitive to pressure increases. These plants exhibit rapid growth capable of quickly growing up through covering layers of silt. According to McCombie and Wile (1971), *Elodea* occurred only at the upper end of the transparency range between 2.7 and 5.8 m. Near the surface the plants were bushy in appearance while at greater depths the internodes were longer.

If not too intense, excessive turbidity and stains are often remedied by planting "mail-forming" species such as *Elodea canadensis*. Mitchell (1974) found plant cover dense in chalk streams. In sparsely populated areas, growth consisted of fibrous roots or matted stems.

Current Wind and Wave Action. Mitchell (1974, p. 92) stated "Stabilization or diminuation of the water flow, control of the water-level fluctuation and ensured continuity of water supply are all factors which potentially promote growth of certain aquatic plants such as *E. canadensis*." Mitchell also claimed that the above factors may have affected water turbidity and depth thus altering light quality such that growth of submerged plants was promoted. Desirable growth conditions for *Elodea* included slow current velocity and a silted bed.

Nutrient Response. *Elodea* tends to prefer high concentrations of nitrogen and phosphorous. Adams et al. (1971) showed that in nutrient enriched waters an increase in nitrogen concentration was observed and correlated with increases in phosphorous, potassium, calcium, manganese, copper, boron, and zinc levels and tended to increase stem diameter. Morphological deviance was noted in nutrient enriched areas; however, *Elodea* grew faster, larger and more luxuriantly with increasing nutrient enrichment. Ryan et al. (1972) also found fertilization or nutrient enrichment to enhance growth unless the growth of algae became competitive. However, vegetative pleomorphism was a consequence of this increased biomass.

Elodea can tolerate a SO_4^{2-} range of 0 to 37 mg/l; pH of 7.0 to 8.8; and alkalinity range as HCO_3^- of 43 to 363 mg/l. The bicarbonate ion can be used as a source of carbon in photosynthesis (Hutchinson 1975).

Susceptibility. Sutton et al. (1969) found treatment of *Elodea* with simazine (80 percent wettable powder formulation) at concentrations of 0, 0.12, 0.50, and 1.00 ppm resulted in a decrease in dissolved oxygen relative to the increase in herbicide concentration. Simazine has been found to cause a decrease in photosynthesis and possibly some alteration in respiration.

Way et al. (1970)applied paraquat in a concentration of 0.5 mg/l which resulted in total elimination of *Elodea*. Diquat in 10 mg/l doses completely killed *Elodea* shoots within six hours. The effectiveness of paraquat or diquat was hindered by the presence of adsorptive organic or inorganic surfaces in the water due to the high affinity of the chemicals for adsorption. These chemicals were most effective during the period fo rapid growth in the summer because *Elodea*, when dormant, was found to be more resistant to the effects of herbicides.

A constant concentration of about 0.25 ppm copper in a stream has been found to eliminate *Elodea* (McIntosh 1974). Control was termed satisfactory with 3 ppm copper sulfate but the effect was temporary.

Productivity

Owens et al. (1967) determined from data on the dry matter and organic carbon contents of *Elodea* that the average growth rate was 0.47 g organic carbon/m^2/day. According to these values, *Elodea* had a 0.12 percent efficiency of utilization of the shortwave solar radiation (3,000 to 30,000 Å)that fell on the lake surface.

Consumer Utilization

Martin and Uhler (1939) considered *Elodea canadensis* to be of little value to waterfowl, since it rarely produced seeds. Consequently, they rated *Elodea* as a "salad" course for captive or other grain-fed ducks.

Elodea canadenis is generally unpalatable to aquatic insects. Hutchinson (1975) suggested that this may be due to chemical protection. Hutchinson further discussed work related to a possible suppression of phytoplankton by *Elodea*. This could be due to an inhibitory substance produced by *Elodea* or possibly competition for light and nutrients.

There appears to be no inhibition of epiphytes by *Elodea*. Epiphytes have been found to grow abundantly between the teeth on the leaf margins and on the upper leaf surfaces (Hutchinson 1975). Bownik (1970) found diverse examples of periphyton in association with *Elodea*. Ramsey (1974) found bacterial assocations to increase seasonally and be more abundant on mature leaves.

Economic Problems and Uses

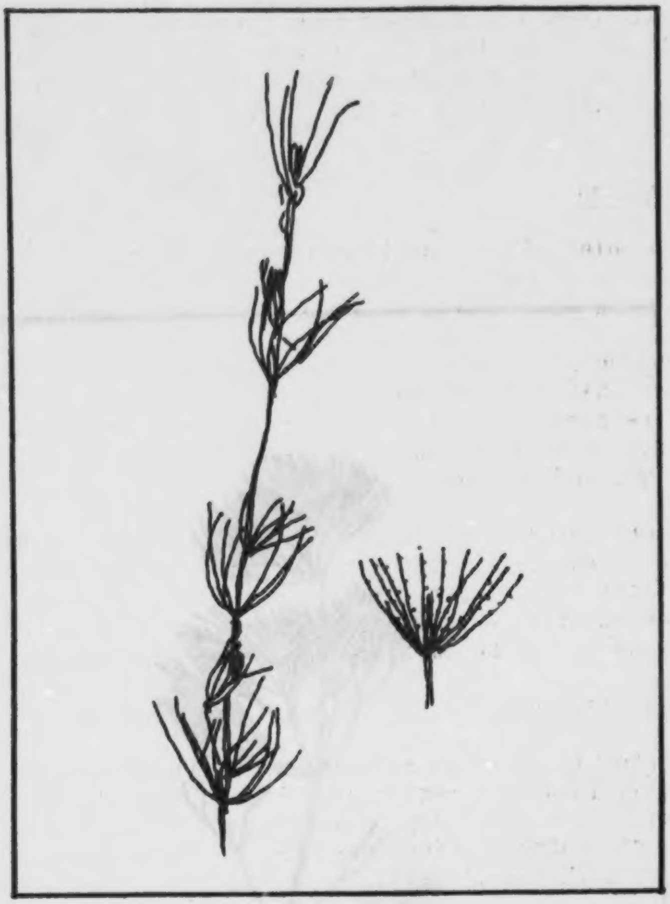
Elodea canadensis is known as "waterweed" and well deserves its name. It characteristically invades a region and often chokes the water system with its prolific growth. The grasscarp or white amur (*Ctenopharygodon idella*), native to large rivers of southern China, has been considered one of the more promising biological controls (Mitchell 1974).

CHARA, *NITELLA* AND *TOLYPELLAS*

Biology

General Vegetative Morphology. The three genera, *Chara*, *Nitella* and *Tolpellas* are members of the single living Characeae family of the single order Charales. Although the Charales differ widely from other algae, the order is commonly grouped in the class Chlorophyceae (Fritsch 1965). The Characeae are commonly referred to as muskgrass or stoneworts. The latter name derives from the stony, brittle texture of the alga caused by lime deposits that are frequently found on the plant in highly calcareous waters (Reid 1961).

Members of the Characeae are macroscopic submerged aquatics with upright green stems divided into multicellular nodes (Hutchinson 1975) (see Figure 25 and 26). All species exhibit a whorled arrangement of leaves arising from special nodes (Fritsch 1965). Rooted by branched rhizomes, species vary in



(copied from Hotchkiss 1967)

Figure 25. Muskgrass (Chara sp.)

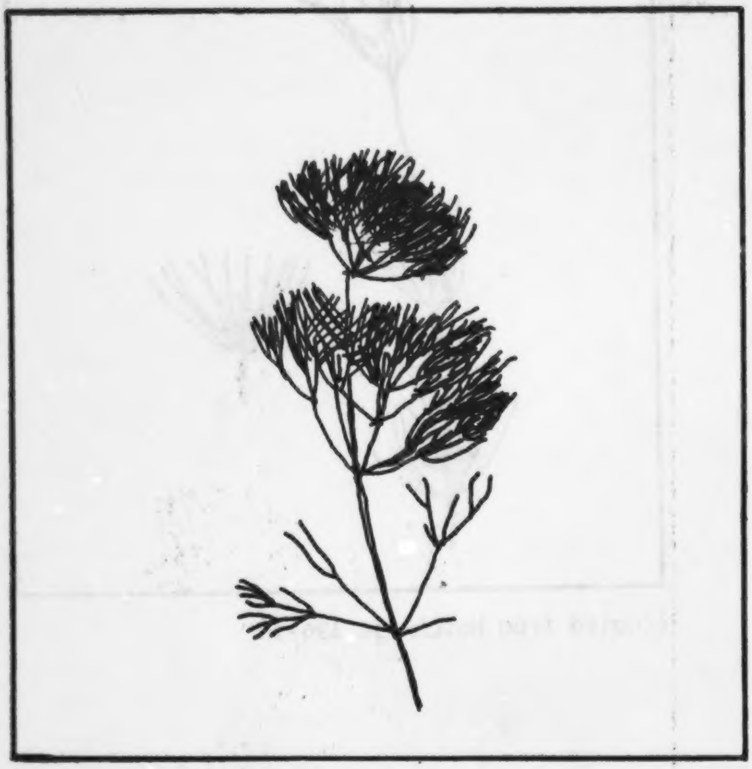


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Figure 26. Nitella (Nitellas sp.)

size from 3 cm to 1.25 m tall. Many species form structures called bulbils (Hutchinson 1975). Found on the rhizome, bulbils serve as anchors as well as reproductive structures. Chara spp. are distinguished from the genera Nitella and Tolypell mainly by the morphology of the apical crown on the oogonium (Hutchinson 1975).

Reproduction. Characeae have highly specialized reproductive organization (Fritsch 1965). Sexual reproduction usually occurs in the summer and autumn, the photoperiod determining the ripening of the antheridia and oogonia (Migula 1909 cited in Hutchinson 1975). The germination of the oospores generally requires from one to three months diapause in cold water (Proctor 1960; Forsberg 1964, cited in Hutchinson 1975). Both sex organs are usually borne on secondary leaves, originating from the nodes, with the oogonium directly above the antheridium (Fritsch 1965). Large numbers of motile sperm cells are produced by the antheridia, fertilizing the oogonia through an opening at the top and producing a resting spore (Schuette and Alder 1929a).

Chara reproduces asexually from bulbils and protonemata that arise from the oospore and later from stem nodes (Hutchinson 1975). Hutchinson suggests that asexual reproduction plays a major role in the species maintenance of the Characeae.

Distribution

The Characeae constitute a group of primarily freshwater macrophytes that forms a substantial part of the world's submerged vegetation in lakes, ponds and streams (Cook et al. 1974). Chara species tend to inhabit hard or calcareous waters and Nitella species generally inhabit soft or circumneutral water (Hutchinson 1975). Inland saline lakes and ponds may be dominated by the Characeae (Ungar 1974). Some species inhabit brackish waters but are not found in truly marine environments (Hutchinson 1975) with various species found in temperate and tropical regions of all the continents (Cook et al. 1974).

Since 1971, MBHRL has conducted an annual summer sampling program for submerged aquatics in the Maryland portion of the Chesapeake Bay. Out of over 600 stations sampled each year, the survey has documented Characeae only in the Susquehanna Flats and Magothy, Severn and Chester Rivers (see Figure 27). However, compared to Potamogeton spp. for example, Chara spp. showed a very low rate of occurrence within the Maryland Bay portion. Chapter 3 discusses available historic data on the Characeae in the Chesapeake Bay.

Environmental Factors Affecting Distribution

Temperature. Temperature affects Characeae distribution through the regulation of oospore germination. Maintenance at about 40 C for one to three months was shown to be necessary to produce germination (Hutchinson 1975). Various species within the Characeae persist throughout the year in temperate climates and can even be found as green plants well below the ice layer (Flossner 1964, cited in Hutchinson 1975).

Salinity. Though Characeae are predominantly freshwater inhabitants, Dawson (1966) allocated about 13 percent of the known species to brackish waters

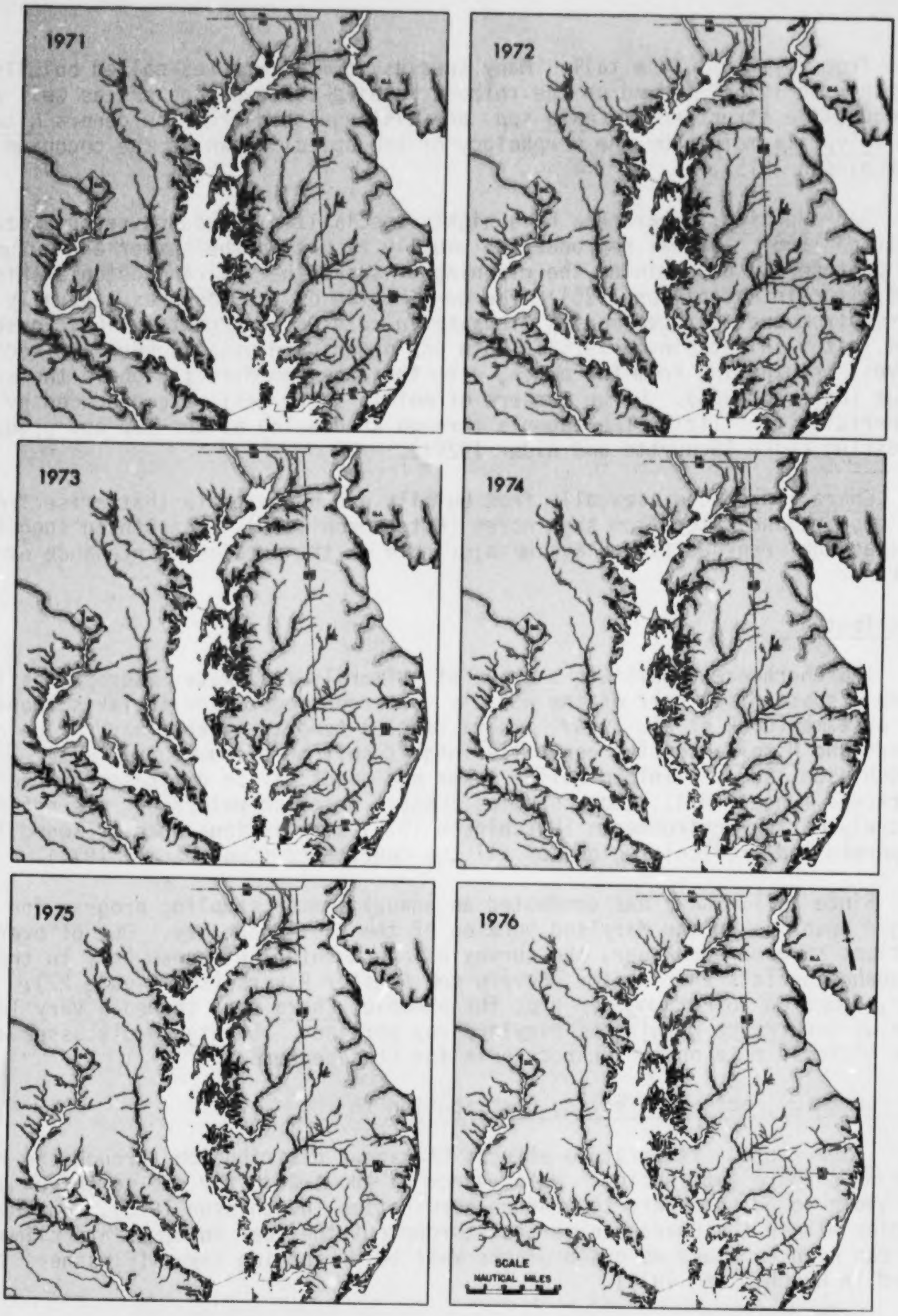
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Figure 27. Distribution of Chara sp., Maryland Chesapeake Bay, 1971-1976

of coastal lagoons, ponds, coves and inland salt lakes and springs. Species ranged in salinities up to 15 ppt with growth cessation and limited survival at 20 ppt.

Substrate. Most species of Characeae grow in silt or mud substrate though a small number of species tend to grow in shallow water on sandy bottoms (Hutchinson 1975). Studies in the Soviet Union (Zenkevitch 1963) uncovered extensive Chareal growths on the shallow hydrogen sulfide silt soils of the eastern coast of Russia. Further abundant growths have been documented in the bays and inlets of the Soviet Aral Sea, growing on black ooze which smelled of H_2S.

Some species require not more than a few millimeters of soft deposits over bedrock to insure survival (Hutchinson 1975).

Light, Depth and Turbidity. The Characeae are capable of surviving in low light intensities, though none of the species are confined to such light intensities in nature. Chara and Nitella species have often been found to grow covered with calcium carbonate or mud deposits, respectively, which provided an effective screen against solar radiation (Hutchinson 1975).

Characeae species can be found inhabiting transparent water well below the limit of flowering plants. The greatest established depth for Chara spp. was in Lake Tahoe at 65.5 m with incident radiation slightly more than 2 percent of that reaching the lake surface (Hutchinson 1975).

The Charales are capable of existing with a small oxygen supply but require pure water; they cannot tolerate turbid or contaminated conditions (Fritsch 1965).

Current Wind and Wave Action. Most of the Characeae are not conspicuous littoral inhabitants, occupying instead the deep waters in lakes and sometimes running water (Hutchinson 1975).

Nutrient Response. Charophyceae are able to take up HCO_3^- as an alternate carbon source to CO_2 (Hutchinson 1975). Chara is known for its ability to inhabit hard water. This may be related to the generally higher availability of HCO_3^- in hard water and the ability of Chara to use the bicarbonate ion.

Anderson (1968) and Forsberg (1964, 1965), working with several different species of Chara, determined that small excesses of phosphorus in the medium inhibited growth. Experiments resulted in plant reduction but not destruction. Optimum development occurred within a range of 5 to 20 mg total P/m^{-3}.

As mentioned previously, Characeae can tolerate large amounts of free H_2S in bottom sediments. The green parts, however, are not as tolerant as are the rhizomes, surviving for two weeks or more at only about 10 percent the H_2S tolerance level of the rhizomes (Hutchinson 1975).

Chara species have a high tolerance for photosynthetic $CaCO_3$ precipitates (Hutchinson 1975). Calcium is essential to Chara, though requirements vary for different species (Fritsch 1965).

Susceptibility. Experiments using the herbicides simazine, propazine and atrazine in whole ponds in Missouri (Walker 1964) determined varied toxicity results for Chara vulgaris. Propazine was the least effective herbicide, needing a maximum application of 3.0 ppm (wettable powder) for only limited control. Simazine and atrazine were slow to provide results. Simazine was applied at rates ranging from 0.5 to 10.0 ppm and 5.5 to 132 kg/ha. Dry applications resulted in zero control. Wet applications resulted in increasing control with increasing concentration of herbicide, the maximum control being achieved at 10.0 ppm. Atrazine applications, corresponding to the simazine rates, resulted in zero control of C. vulgaris. Applications of herbicides during cold, cloudy weather periods indicated that such weather did not favor the efficacy of the chemicals. However, some limited growth inhibition did result from cold weather applications.

Experimental work performed in Great Britain (Newbold 1975) recorded the effects of eight herbicides approved by the Pesticide Safety Precautions Scheme for use in or near water. Dichlobenil and chlorthiamid were found to be the most effective, resulting in a 2-year kill of Chara spp. Terbutryne resulted in a one-year kill and diquat and paraquat resulted in an initial kill, but revegetation occurred within the same year. Applications of dalapon, dalapon-paraquat and 2,4-D were not recommended for use and considered likely to be inefficient.

Consumer Utilization

The Characeae are of considerable economic value and have been used for water purification, as food for farm stock and fish stock, as an agent in settling silt and in the manufacture of polishes (Cook et al. 1974). Many species of ducks feed exclusively on Charophyceae while some species feed solely on the bulbils (Hutchinson 1975).

By forming often times vast bottom meadows, the Characeae provide habitat for aquatic fauna plus support for many lesser epiphytes (Needham and Lloyd 1930). According to Fassett (1960), Chara provides fair shelter and an excellent source of fish food, especially for juvenile trout and large mouth and small black bass.

Martin and Uhler (1939) studied aquatic plant usage by game ducks in various areas of the United States and Canada. Based on volumetric percentage of stomach materials, the study rated Characeae as having good to excellent usage, the plants being consumed by many kinds of waterfowl, especially diving ducks. All parts of the plants were discovered in duck stomachs, especially the oogonia and bulbils.

Cottam (1939) studied the contents of gizzards and gullets from approximately 3,500 inland plant-feeding ducks in North America. Along with the pondweed family, Chara spp. was included in the list of submerged aquatics having greatest value to Nyroca spp. and ruddy ducks. The study revealed that the ducks were utilizing all parts of the plants: seeds, rhizomes, leaves and bilbils.

Economic Uses and Problems

Impoundments. Chara has been utilized in southeast U.S. as waterfowl food in shallow fresh water lakes and ponds (Ball 1965). However, beneficial results were found to be temporary unless extensive management procedures were used to prevent natural plant succession and invasion of pest plants.

Impacts to Mosquitoes. Hutchinson (1975) cited experimental work performed in numerous countries on the effects of Chara and Nitella on mosquitoes. Experimental evidence implicated various Chara species as having possible larvicidal properties and egg laying inhibitors. Hutchinson expressed the need for experimental determination of the possibility of using native species of Chara and Nitella for mosquito control.

CERATOPHYLLUM DEMERSUM

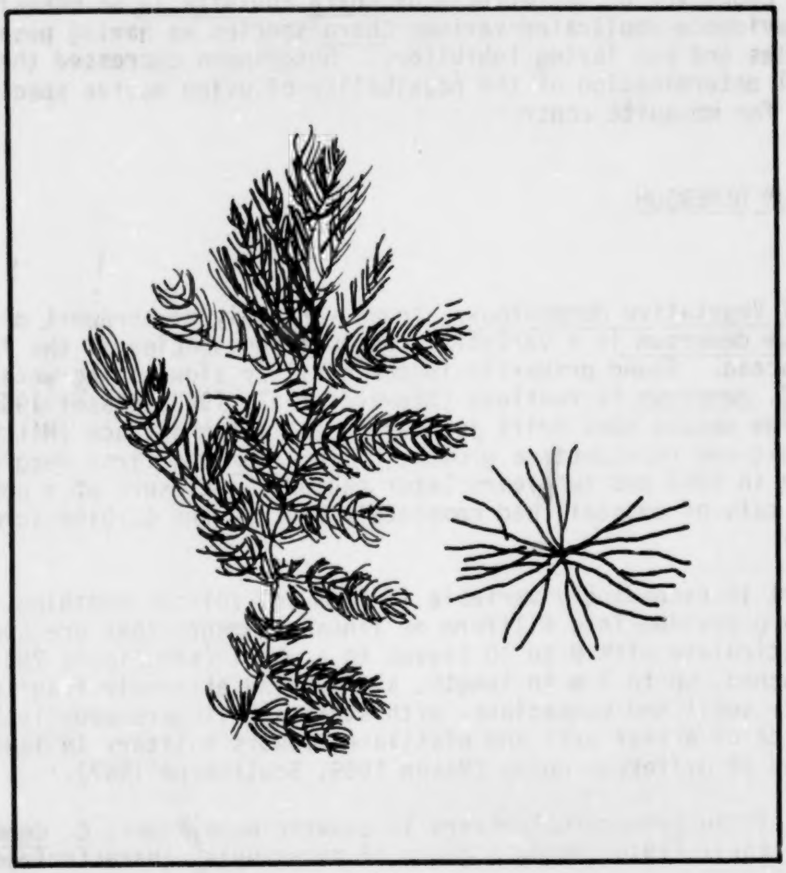
Biology

General Vegetative Morphology. Known variously as hornwort or coontail, Ceratophyllum demersum is a variable, cosmopolitan species of the family Ceratophyllaceae. Found primarily in the still or slow moving waters of streams and ponds, C. demersum is rootless (Steward et al. 1960; Fasset 1960). Coontail may form large masses that drift just below the water surface (Mitchell 1974). Capable of extreme reproductive growth, C. demersum was first recorded in a New Zealand lake in 1963 and two years later caused the closure of a power station after large mats of coontail had completely blocked the turbine screens (Chapman et al. 1974).

Hornwort is exceedingly variable in form and foliage toothing. In general, the leaves are divided into filiform or linear segments that are sparsely forked, minutely denticulate with 9 to 10 leaves to a whorl (see Figure 28). Stems are densely branched, up to 3 m in length, slender and extremely fragile. Flowers are generally small and monoecious, with staminate flowers usually borne in pairs on either side of a leaf axil and pistilate flowers solitary in leaf axial, both usually found at different nodes (Mason 1969; Sculthorpe 1967).

As one of the primary colonizers in aquatic ecosystems, C. demersum is included by Mitchell (1974) among a group of macrophytes characterized by the following:

- capable of rapid vegetative growth;
- able to regenerate from small vegetative portions;
- partially independent from structures of sexual reproduction;
- attains large areas of photosynthetic tissue through vegetative growth; and
- independent of water level fluctuations and bottom substrate.



(copied from Hotchkiss 1967)

Figure 28. Coontail (*Ceratophyllum demersum*)

Reproduction. *Ceratophyllum demersum* reproduces mainly through vegetative means. Almost any fragment of stem bearing a lateral bud is capable of forming a new plant. Sexual reproduction occurs when the mature anther reaches the surface, dehisces and releases microspores which sink down slowly to the female flower (Sculthorpe 1967). Fertilization produces a one-seeded nutlet which needs to remain in the water in order to germinate. Experiments by Muenscher (1936) indicated that once seeds were dried, no germination would occur.

Distribution

Ceratophyllum demersum is considered to be truly cosmopolitan exhibiting remarkable latitude penetration. Fossil records show hornwort to be well represented in Pliocene deposits at Castle Eden and Palsefield in Suffolk (Sculthorpe 1967). Worldwide, coontail frequents quiet, fresh water pools and slow streams (Mason 1969). Fragments of *Ceratophyllum* are frequently transported casually when caught in waterfowl feathers (Sculthorpe 1967).

MBHRL Survey work from 1971 to 1976 has documented *C. demersum* as sparse and limited primarily to the Susquehanna Flats, Magothy, Severn and Chester Rivers (see Figure 29). Chapter 3 includes available data by river system pertaining to historic distribution of hornwort in the Maryland portion of the Chesapeake Bay.

Environmental Factors Effecting Distribution

Temperature. Fruit maturation of *C. demersum* seems to require almost tropical temperature conditions, thus limiting sexual reproduction mainly to warm climates (Martin and Uhler 1939). Wilkinson (1963) determined a critical minimum temperature for vegetative growth of 20 C with optimum growth at 30 C.

Salinity. *Ceratophyllum demersum* is essentially a fresh water macrophyte (Martin and Uhler 1939). Salinity experiments (Bourn 1932) conducted under favorable soil, CO_2, light and temperature conditions resulted in growth retardation of *Ceratophyllum* with increasing concentrations of salinity. Based on percentage increase in dry weight, *Ceratophyllum* decreased in growth roughly proportional to the increase in salinity. However, plants appeared to develop normally in salinities under 6.5 ppt. Above 6.5 ppt, plants showed size reduction, spindly stems and curling leaves. An endurance limit was established at about 8 ppt.

Substrate. Hornwort often grows independently of substrate material. According to experiments by Shannon (1953), *Ceratophyllum* showed no capability for root production even when grown in wet sand or wet sphagnum. However, when *C. demersum* plants were suspended over sand or mud, the plants grown over mud produced a mean crop that was twice that produced over sand (Hutchinson 1975).

Though available literature refers to *Ceratophyllum* as being rootless (Sculthorpe 1967), there appears to be some question as to whether *C. demersum* is always rootless in the Cheapeake Bay. Stotts (personal communication) has found hornwort in the Bay that has been rooted. This may be due to the location of the Bay in a temperature transition zone for *Ceratophyllum*. The Bay has both

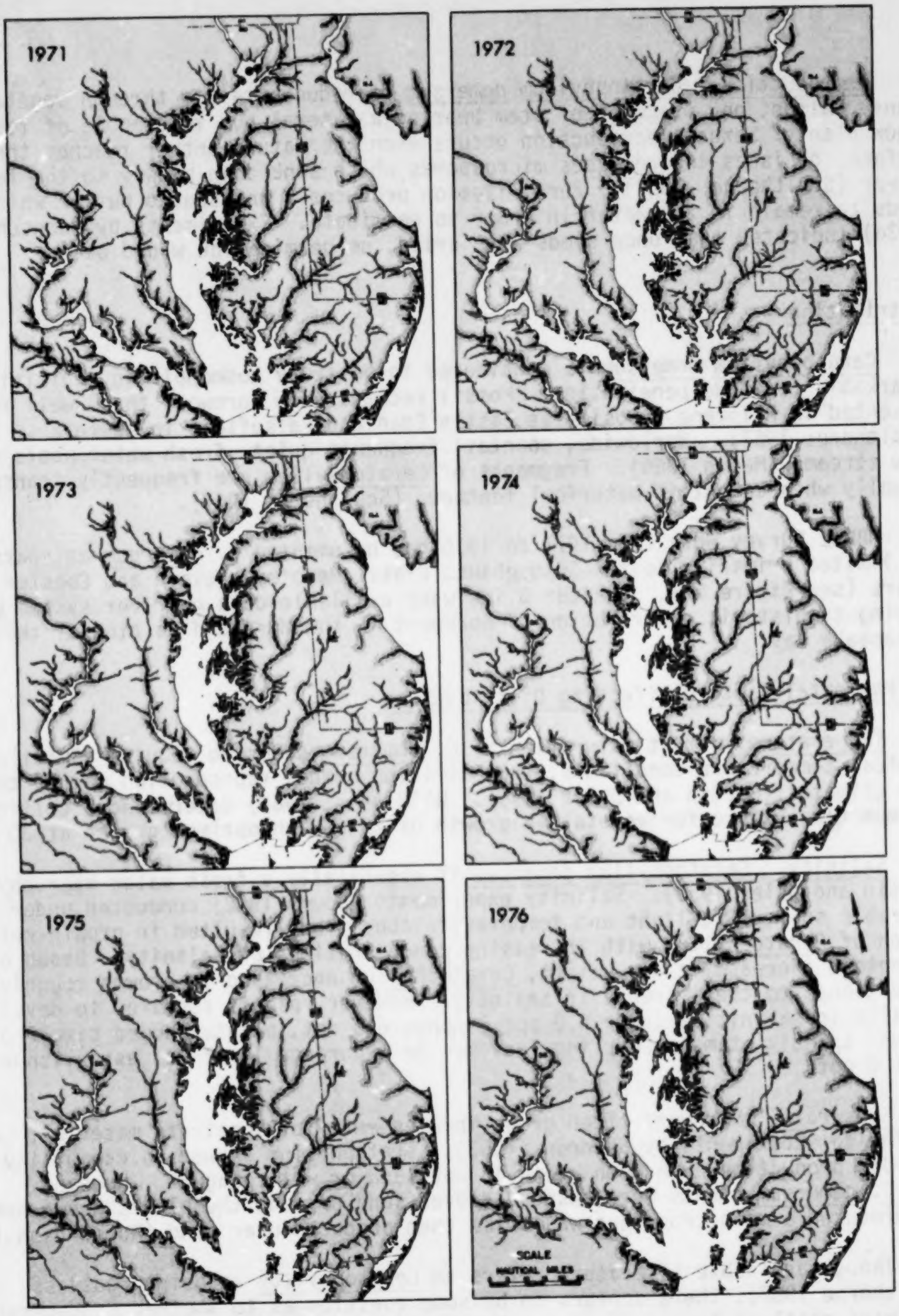


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Figure 29. Distribution of Ceratophyllum demersum, Maryland Chesapeake Bay, 1971-1976

sheltered areas with higher ambient temperatures possibly more conducive to sexual reproduction and more exposed areas with lower mean temperatures where vegetative growth could be favored (Confer, personal communication).

Light, Depth and Turbidity. Coontail is shade tolerant (Lamoureux 1957) requiring a minimum of 2 percent full sunlight for optimum growth (Chapman et al. 1974). Experiments conducted in an Ohio Lake (Meyer et al. 1943) concluded that *C. demersum* reached a compensation point of zero photosynthesis at less 2 percent full sunlight intensity. In Back Bay and Currituck Sound, *C. demersum* was found thriving in the dark brown water draining from the swamps (Bourn 1932). Coontail is sufficiently shade tolerant to exist under floating leaved plants (Lind and Cottam 1969).

Ceratophyllum generally is not considered to be depth limited due to its often rootless nature. Turbidity is not as negative a factor for coontail as for rooted aquatics due to shade tolerance and water surface habitat.

Current Wind and Wave Action. *Ceratophyllum demersum* is sensitive to current,wind and wave action (Mitchell 1974). Any violent disturbance has been found to break the fragile stems (Sculthorpe 1967). However, this does not result in mortality but instead acts in dispersal and increased vegetative growth.

Nutrient Response. *Ceratophyllum* absorbs necessary nutrients from water rather than soil media (Sculthorpe 1967). Toetz (1973) suggested that for *Ceratophyllum*, dissolved NH_4 input from sediments, animal excretion, autolysis and microbial decomposition might have been the ultimate limiting factor in waters with an immense biomass. Table 13 lists several other elements found to concentrate in *Ceratophyllum*.

Table 13. Concentration factors for *Ceratophyllum demersum*[a]

Element	Mean concentration water (mg/l^{-1})	Mean concentration dry plant (ppm x 0.1)	Concentration factor
Cooper	.0160 .0083	3.23 1.52	202 183
Zinc	.0090	16.40	1820
Cadmium	.0015[b]	9.12[b]	600 to 700

[a] Hutchinson 1975

[b] Value in ppm

Susceptibility. Ceratophyllum has been shown to be highly resistant to large concentrations of boron. Boron is even stimulatory to photosynthesis within a range of 0.5 to 100 mg/l. Greater concentrations were found to affect protein synthesis, water balance and nutrient assimilation (Wetzel 1964). In normal water, arsenic has been found to accumulate in C. demersum up to 26 ppm. In geothermal waters in New Zealand coontail has been recorded to contain 1,000 ppm arsenic as dry material (Hutchinson 1975).

Attempts to eradicate Certophyllum demersum have been well documented. Table 14 lists various herbicides and related application rates and control results.

Various fauna are known to consume C. demersum including Chinese grass carp or white amur (Ctenopharyngodon idella). The Brazilian snail (Marisa cornvarietis) also consumes coontail; however, research into the use of these snails for control of Ceratophyllum has shown that such action would probably disturb the entire ecosystem (Chapman et al. 1974).

Productivity

As has been previously mentioned, C. demersum is highly productive through vegetative means. Any fragment that includes a bud is capable of producing a new plant. Chapman et al. (1974) found high photosynthesis rates in these apical buds plus in relatively new leaves. Stems and older leaves produced considerably lower amounts of oxygen.

Consumer Utilization

Ceratophyllum demersum is not unanimously considered to be of great importance to wildlife. Cottam (1939) did not include coontail in his listing of aquatic macrophytes having great value to Nyroca and ruddy ducks. Lamoureux (1957) rated Ceratophyllum as slightly important as duck food with seeds and leaves eaten only occasionally. Sculthorpe (1967), however, rated foliage and seeds as having great food importance to ducks, coots, geese, grebes, swans, waders, shore and game birds. Ceratophyllum foliage has further moderate importance as fish food. As a shade, shelter and spawning medium, Sculthorpe rated coontail high. The dissected leaves of Ceratophyllum may support dense communities of epiphytic desmids, diatoms, filamentous algae and zooplankton which all provide food for fish.

On a somewhat different note, preparations of leaves have historically been used internally as purgatives and diuretics, for dysentery and as remedies for bilousness and jaundice. Topically, leaf extracts have been used to treat anything from elephantiasis to sunburn (Sculthorpe, 1967).

Table 14. Effects of certain herbicides on Ceratophyllum demersum

Herbicide	Application rate	Control
Silvex, K salt	1.5 to 2 ppm	100%
	22 kg/ha granules	100% initial kill with regrowth
Diquat dibromide	1 to 2.5 ppm	75 - 80% after 24 hours
	5 to 10 ppm	89 - 100% after 24 hours
Diquat	5 ppm	Satisfactory
Paraquat	1 to 2.5 ppm	50 - 65% after 24 hours
	5 to 10 ppm	90 - 97% after 24 hours
Ipazine	1 to 2.5 ppm	0 - 5% after 24 hours
	5 to 10 ppm	10% after 24 hours
Acrolein	1 to 2.5 ppm	79 - 95% after 24 hours
	5 to 10 ppm	99 - 100% after 24 hours
Endothall	1 ppm	Poor
	2 to 3 ppm	For Season
	16.5 to 33 kg/ha granules	None
	5 ppm granules	Effective
Endothall DDS	1 to 5 ppm	Good
2,4,-D	3 ppm	Good
	33 kg/ha granules	Good
Ametryne	4 to 5 ppm	Good
Sodium arsenite	1 to 2 ppm	90%
	7 to 10 ppm	Killed within one week
	10 to 20 ppm bottom granules	Unsatisfactory
Simazine	5 to 6 ppm wettable powder	Erratic
	6 ppm granules	Satisfactory
Atrazine	5 to 6 ppm wettable powder	Erratic
	5 to 6 ppm granules	Satisfactory

Source: Lawrence and Hollingsworth 1969.

NAJAS GUADALUPENSIS

Biology

General Vegetative Morphology. Najas guadalupensis is commonly referred to as bushy pondweed (Radford et al. 1964) or water nymph (Schuette and Alder 1929b). Physically resembling Zannichellia and Elodea, Najas is the single genus of the Najadaceae family. Members of the genus are either monoecious or dioecious, annual or perennial and have slender branched stems with simple, linear, opposite leaves that are often marginally toothed and have sheathed bases (See Figure 30) (Cook et al. 1974).

Around the turn of the century there appears to have been a rather heated debate among several reknowned botanists regarding the primitive or advanced nature of Najas. Rendle (1899, cited in Arbor 1920) maintained that Najas was a primitive monocotyledon. Guppy's "Differentiation Theory" also categorized Najas as a primitive species. He maintained that N. marina had evolved into other distinct species within the same genus since N. marina exhibited a wider range than other species of Najas. Arbor (1920) disagreed with both Rendle and Guppy, stating that the genus Najas was a "highly reduced form representing perhaps the ultimate term of reduction in the Potamogetonaceae series." Schuette and Alder (1929), some years after the previous controversy, referred to Najas as appearing to be a primitive monocotyledon.

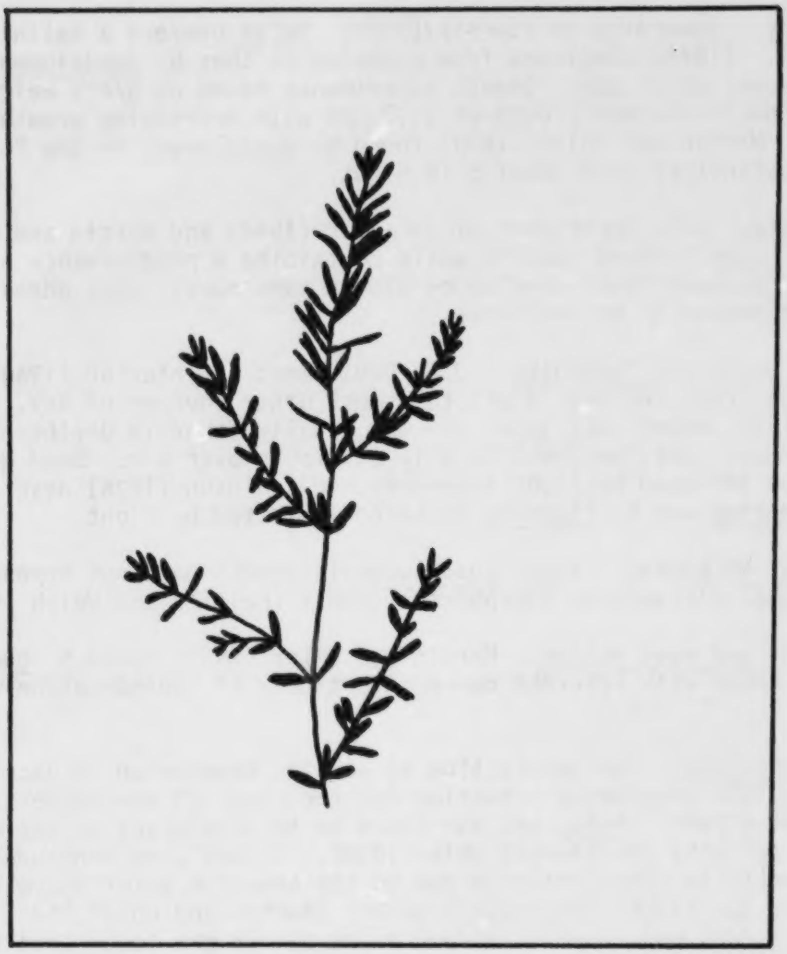
There appears to be further confusion in literature regarding whether Najas spp. are annual or perennial. Radford et al. (1964) and Hutchinson (1975) refer to the genus as annual. Arbor (1920), however, cites only the species N. minor and N. flexibis as being annuals. Cook et al. (1974) leaves the choice open for individual species and simply lists the genus as either annual or perennial. Information as to which group N. guadalupensis fits in is not readily available.

Najas spp. do not have rhizomes or tubers as do most other species of SAV (U.S. Department of the Interior 1944). Roots are simple and small with a radicle that is either short-lived or undeveloped (Arbor 1920; Bourn 1932).

Reproduction. Najas spp. appear to reproduce primarily sexually. Pollination takes place underwater (Hutchinson 1975) as the globular or elipsoid microspores float up through the water and are caught on the elongated stigmas (Schulthorpe 1967). The female flower consists of a single ovule that sometimes is surrounded by one or two envelopes, evidently depending on the species (Arbor 1920).

Vegetative reproduction does not appear to be as common or successful as sexual reproduction. In discussing attempts to propagate Najas sp., the U.S. Department of Interior (1944) suggested propagation by whole plants or seeds. Vegetation by cuttings seemed to require a specimen with at least three nodes.

Distribution. Najas is essentially a freshwater species that ranges in habitat from Oregon to Quebec and from California to Florida (Hotchkiss 1967). Martin and Uhler (1939) considered Najas guadalupensis as a fresh or mildly brackish water species.



(redrawn after Hotchkiss 1967)

Figure 30. Naiad (Najas sp.)

Distribution of *N. guadalupensis* in the Maryland portion of the Chesapeake Bay presently appears to be limited to the Chester River (see Figure 31). Historically, *Najas* was once abundant on the Susquehanna Flats. The Susquehanna Flats Survey (1958 to 1975) documented *Najas* through 1971. Then *Najas* suddenly disappeared and neither the Susquehanna Flats or MBHRL survey teams found *Najas* in the area until 1977 when the Flats survey teams documented *N. guadalupensis* as "rare" (Stotts, personal communication).

Environmental Factors Affecting Distribution

Salinity. According to Steenis (1970), *Najas* prefers a salinity of 3 ppt. Haller et al. (1974) concluded from experiments that *N. guadalupensis* died after four weeks at 10 ppt. Growth experiments based on g/dry weight measurements resulted in maximum growth at 0.17 ppt with decreasing growth in increasing salinities. Martin and Uhler (1939) found *N. guadalupens* in the Potomac River growing in salinities from about 6 to 9 ppt.

Substrate. U.S. Department of Interior (1944) and Martin and Uhler (1939) agreed that *Najas* thrived best in soils containing a predominance of sand though the species has been found growing on almost pure muck. USDI added that firmer substrates appeared to be preferred.

Light, Depth and Turbidity. U.S. Department of Interior (1944) stated that *Najas* sp. required less light than most other species of SAV. Martin and Uhler (1939) added that *Najas* sp. was usually found in depths ranging from 0.3 to 1.2 m but could be found locally at depths over 6 m. Seed germination appears to be affected by light intensity. Hutchinson (1975) described germination of *N. marina* and *N. flexilis* as being inhibited by light.

Nutrient Responses. *Najas guadalupensis* growth does not appear to be related to either nitrogen or phosphorous levels (Peltier and Welch 1969).

Currents and Wave Action. Martin and Uhler (1939) found *N. guadalupensis* growing in areas with retarded currents but only if sedimentation was not extreme.

Susceptibility. The destruction of aquatic vegetation in Back Bay/Currituck Sound around 1929 provided a situation for the study of environmental factors affecting SAV growth. *Najas* sp. was found to be intolerant of extreme turbidity and varying salinity (Martin and Uhler 1939). It was also concluded that negative impacts to *Najas* occurred due to the brackish water hydroid *Cordylophora lacustris* and the fungus *Rhizoctonia solani* (Martin and Uhler 1939). Crawfish *Cambarus rusticus*) was found to control *Najas* sp. in the lower Ohio River and its tributaries (Martin and Uhler 1939). Smothering by benthic algae was also found to be a successful method of control (Sculthorpe1967). Mitchell (1974) listed the marisa snail (*Schistosoma mansoni*) as providing seasonal control. Grass carp feed on young *Najas* stems and leaves along with *Potamogeton* sp. and *Elodea* sp. though established, mature growth was not controlled. In discussing the grass carp (*Ctenopharyngodon idellus*), the Avauet (1965) concluded that the carp could effect complete control of *N. guadalupensis* after two to three weeks in experimental plastic pools. Seaman and Porterfield (1964) found complete control of *N. guadalupensis* by the fresh water snail *Marisa cornuarietis*.

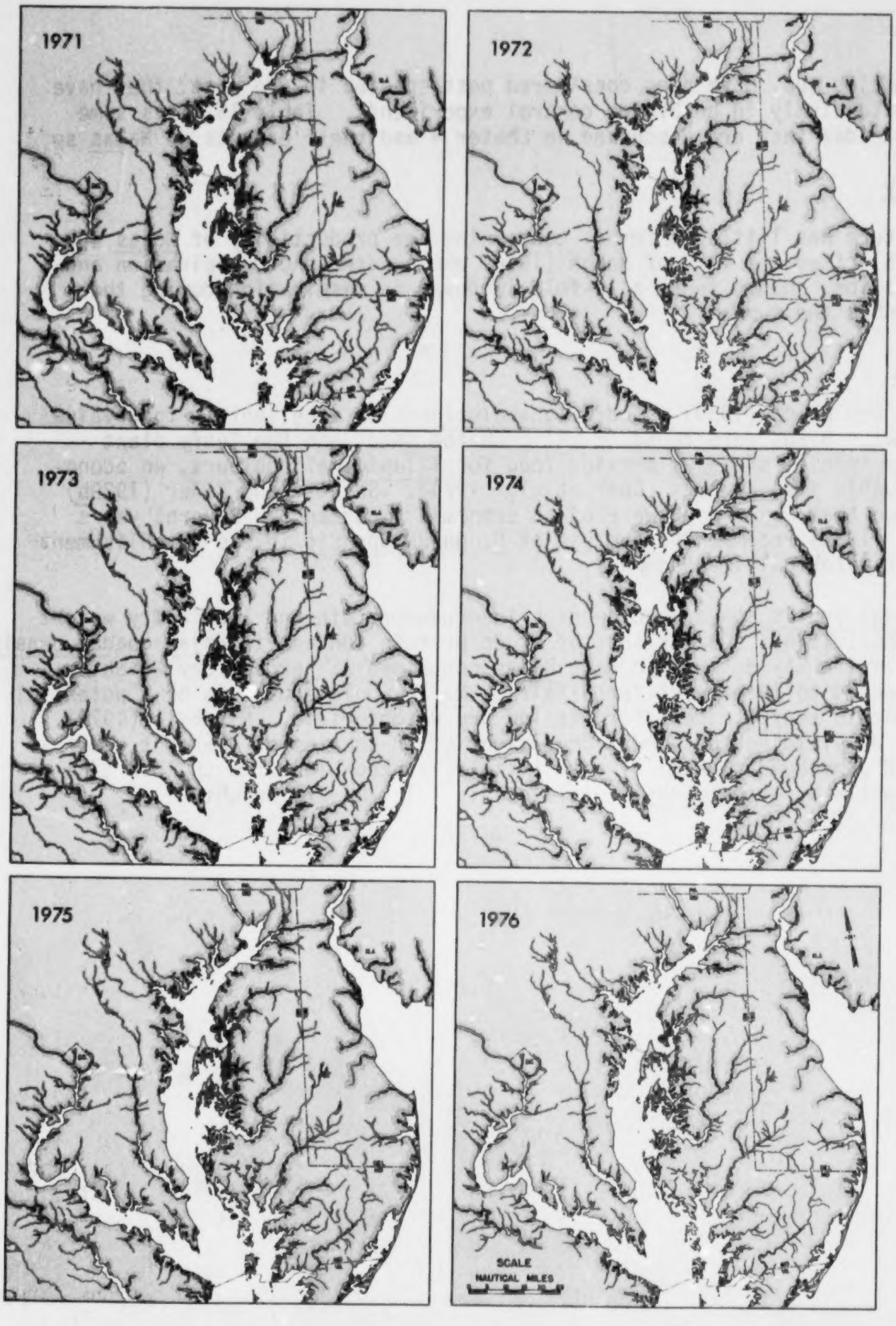

1971
1972
1973
1974
1975
1976
SCALE
NAUTICAL MILES

Figure 31. Distribution of Najas sp., Maryland Chesapeake Bay, 1971-1976

Since Najas spp. have been considered pest species in the past, they have been used extensively in herbicide control experiments. Table 15 lists some of the herbicides that are discussed in Chater 4 and their effects on Najas sp.

Productivity

Literature has little to reveal concerning the productivity of Najas sp. Wetzel (1975) cited the work of Hough (1974) who studied photorespiration and dark respiration. Hough found a 10-fold increase in respiration during the fall compared to the summer.

Utilization

Martin and Uhler (1939) rated N. guadalupensis as excellent in food value for waterfowl. Birds were found to eat both the seeds and the leafy plant parts. Many species of Najas provide food for Tilapia melanopleura, an economically valuable fish species (Cook et al. 1974). Shuette and Alder (1929b) firmly stated that Najas spp. were of no economic importance. Several years later, USDA (1939) recommended the use of N. guadalupensis in the establishment of permanent waterfowl impoundments.

N. guadalupensis appears to be high in crude protein and low in dry weight yield (Mitchell 1974). Comparisons of crude protein content in Najas guadalupensis and young alfalfa hay determined that Najas contained 22 percent dry weight as protein compared to 19 percent for alfalfa. Najas would appear to be a potential valuable protein source except for its low dry weight yield. Mitchell (1974) lists Najas as yielding the lowest dry weight of 18 species of emergent, terrestrial and submerged plants. Typha latifolia ranked highest with 15.3 t/ha dry weight while N. guadalupensis yielded only 1.1 t/ha dry weight.

Table 15. Herbicide control of *Najas* sp.

Herbicide	Application rate	Control	Source
Diquat			
1st year	2.75 kg/ha	complete	Lawrence 1965
+ paraquat 2nd year	1.1 kg/ha	complete	Lawrence 1965
Diquat			
(4 week exposure)	1 ppm	98%	Blackburn 1963
	5 ppm	100%	Blackburn 1963
	10 ppm	100%	Blackburn and Weldon 1964
Diquat			
(16 weeks after exposure)	1 ppm	85%	Blackburn and Weldon 1964
Paraquat			
1st year	2.75 kg/ha	complete	Lawrence 1965
+ paraquat 2nd year	1.1 kg/ha	complete	Lawrence 1965
Paraquat			
(16 weeks after exposure)	1 ppm	90%	Blackburn and Weldon 1964
Paraquat			
(4 week exposure)	1 ppm	87%	Blackburn 1963
	5 ppm	100%	Blackburn 1963
	10 ppm	100%	Blackburn 1963
Fenac			
1st year	11.0 kg/ha	complete	Lawrence 1965
2nd year	none	complete	Lawrence 1965
Diquat			
1st year	0.66 kg/ha	complete	Lawrence 1965
+ paraquat 2nd year	11.0 kg/ha	complete	Lawrence 1965
Simazine			
1st year	11.0 kg/ha	complete	Lawrence 1965
2nd year	none	complete	Lawrence 1965
Diquat and Simazine			
1st year	0.66 kg/ha	complete	Lawrence 1965
	11.0 kg/ha	complete	Lawrence 1965
+ paraquat 2nd year	1.1 kg/ha	complete	Lawrence 1965
Endothal	1 ppm	99%	Blackburn 1963
	5 ppm	100%	Blackburn 1963
	10 ppm	100%	Blackburn 1963
Linuron			
1st year	11.0 kg/ha	complete	Lawrence 1965
2nd year	none	complete	Lawrence 1965
Paraquat and Linuron			
1st year	0.1 kg/ha	complete	Lawrence 1965
	5.5 kg/ha	complete	Lawrence 1965
2nd year	none	complete	Lawrence 1965

CHAPTER 2

THE ROLE OF SUBMERGED AQUATIC VEGETATION

INTRODUCTION

Submerged aquatic vegetation is an integral part of an estuarine system such as the Chesapeake Bay. Submerged macrophytes constitute the principal food source for waterfowl and some fish; they provide, direct or indirect, food and shelter for many of the small host organisms that are eaten by fish and other predators; the spawning activities of certain organisms require them; they purify the water of such noxious substances as carbonic acid and return oxygen; they shade the underlying waters and sediments from solar heating; reduce turbidity by precipitating fine sediments; and provide an important source of detritus. The assimilation of inorganic substances into organic compounds usable by organisms enhances the importance of these plants as a vital link in the food chain. Functioning as prime areas for hiding and breeding, this vegetation group provides surfaces for attachment of eggs. Also, an abundance of animal life, including insects, crustacean, molluscan, fish and others find excellent feeding grounds on SAV (Baker 1916).

Submerged macrophytes help stabilize sediments and reduce shoreline erosion. Bays with benthic flora have been shown to have relatively stable metabolism with less fluctuation in comparison to plankton dominated bays (Odum and Hoskin 1958). SAV further functions in the estuarine nutrient cycle as a nutrient trap for dissolved phosphorus and nitrogen (Clark et al. 1973).

Though not presently utilized for such purposes in the Chesapeake Bay area, aquatic plants serve as livestock food, human food, soil additives, fuel and wastewater treatment in various other parts of the world. Primary utilization in the Chesapeake Bay area is for wildlife conservation.

WATERFOWL

The Chesapeake Bay is an important waterfowl feeding area used by migrating and wintering birds. Migration patterns can be classified according to particular flyways and specific migration routes (Bellrose 1976). Distinguishing a Flyway is a useful tool for interpreting waterfowl population trends since migratory birds show greater affinity to a particular Flyway than to the country as a whole. Flyways cover extensive areas and show population movement from the general north-south direction. Migration routes, however, are more clearly defined since they are 16 km wide at most and birds are directed to river valleys, lakes and sea coasts that are in sight.

The two most important migration paths pertinent to the Bay are the eastern route, primarily through Quebec, and the route from the central and northwest Canadian interior breeding grounds (Lincoln 1953; Bellrose 1976). Some Greater Snow Geese and American Brant, originating as far north as the eastern Artic islands and Greenland coast, partly follow coastal routes to Mid-Atlantic bays and sounds. A more interior route directs Black Ducks and Canada Geese from northern points of Quebec and Ontario.

Most diving ducks of the genus Aythya and a large portion of the dabbling ducks are produced in the north central part of the continent and migrate southward through the Great Lakes region (Bellrose 1976). Thus, a northwest to southeast route is pursued by Canvasback, Redhead, and Greater and Lesser Scaup wintering in Chesapeake Bay, Back Bay and other Mid-Atlantic coastal waters.

For the purpose of this report, waterfowl species inhabiting the Bay area are arranged into genera and represented by nomenclature used by Bellrose (1976) (see Table 16).

A review of literature (Stewart and Manning 1958; Addy 1953; Longwell and Stotts 1958; Stotts 1955; Henny and Holgersen 1974) indicates various waterfowl species in Chesapeake Bay are undergoing changes in distribution, population size and feeding habits. Data sources monitoring such changes include aerial population counts and gullet and gizzard food content analyses. Observations of banded birds are an additional information source used to trace harvest patterns, mortality, migration corridors and breeding ground derivations. Whistling Swan patterns also have been studied by using collar bands which facilitate multiple resighting of birds. Radio telemetry is another tracing method with a normal recovery range of 8 to 240 km (Sladen 1975). About 1,000 Canada Geese are banded each winter from primary Eastern Shore flocks to monitor the status of the population (Stotts, personal communication). Observations of production and band recoveries of diving ducks and also dabbling ducks such as Gadwalls and Black Ducks have indicated changes in the breeding range and migration routes for these species.

Population Trends of Wintering Waterfowl in Chesapeake Bay

Information on the distribution and population count for all waterfowl species of the United States is available from the U.S. Bureau of Sport Fisheries and Wildlife, Migratory Bird and Habitat Research Laboratory, Laurel, Maryland. This annual aerial continental survey (January inventory), initiated in 1948, is presently ongoing in cooperation with State and provincial conservation departments. Graphs of the population trends for eight species of waterfowl inhabiting the Atlantic Flyway and the Chesapeake Bay region are represented in Figure 32.

The two most abundant puddle ducks on Chesapeake Bay are Black Ducks and Mallards. Generally, Black Ducks and Mallards have maintained fairly stable populations after declines from outstanding highs observed in the mid-1950's. The high population of Mallards was a result of excellent water conditions on the western and northern breeding grounds succeeded by poor water conditions in

Table 16. Waterfowl of the Chesapeake Bay[a]

Family Anatidae (Swans, Geese and Ducks)

Subfamily Anserinae (Swans, Geese)

Cygnus columbianus (Whistling Swan)
Cygnus olor (Mute Swan)...recently introduced
Cygnus buccinator (Trumpeter Swan)... formerly present
Anser c. caerulescens (Lesser Snow Goose)
Anser caerulescens atlantica (Greater Snow Goose)
Branta canadensis (Canada Goose)
Branta bernicla hrota (Atlantic Brant)

Subfamily Anatinae (Surface-Feeding and Diving Ducks, Mergansers, Stiff-Tailed Ducks)

Aix sponsa (Wood Duck)
Anas penelope (European Wigeon)... rare
Anas americana (American Wigeon)
Anas strepera (Gadwall)
Anas crecca carolinensis (Green-winged Teal)
Anas platyrhynchos platyrhynchos (Mallard)
Anas rubripes (Black Duck)
Anas acuta acuta (Pintail)
Anas discors (Blue-winged Teal)
Anas clypeata (Northern Shoveler)
Aythya valisineria (Canvasback)
Aythya americana (Redhead)
Aythya collaris (Ring-necked Duck)
Aythya marila mariloides (Greater Scaup)
Aythya affinis (Lesser Scaup)
Clangula hyemalis (Oldsquaw)
Melanitta nigra americana (Black Scoter)
Melanitta perspicillata (Surf Scoter)
Melanitta fusca deglandi (White-winged Scoter)
Bucephala albeola (Bufflehead)
Bucephala clangula americana (Common Goldeneye)
Mergus cucullatus (Hooded Merganser)
Mergus serrator (Reh-breasted Merganser)
Mergus merganser americanus (Common Merganser)
Oxyura jamaicensis ribida (Ruddy Duck)

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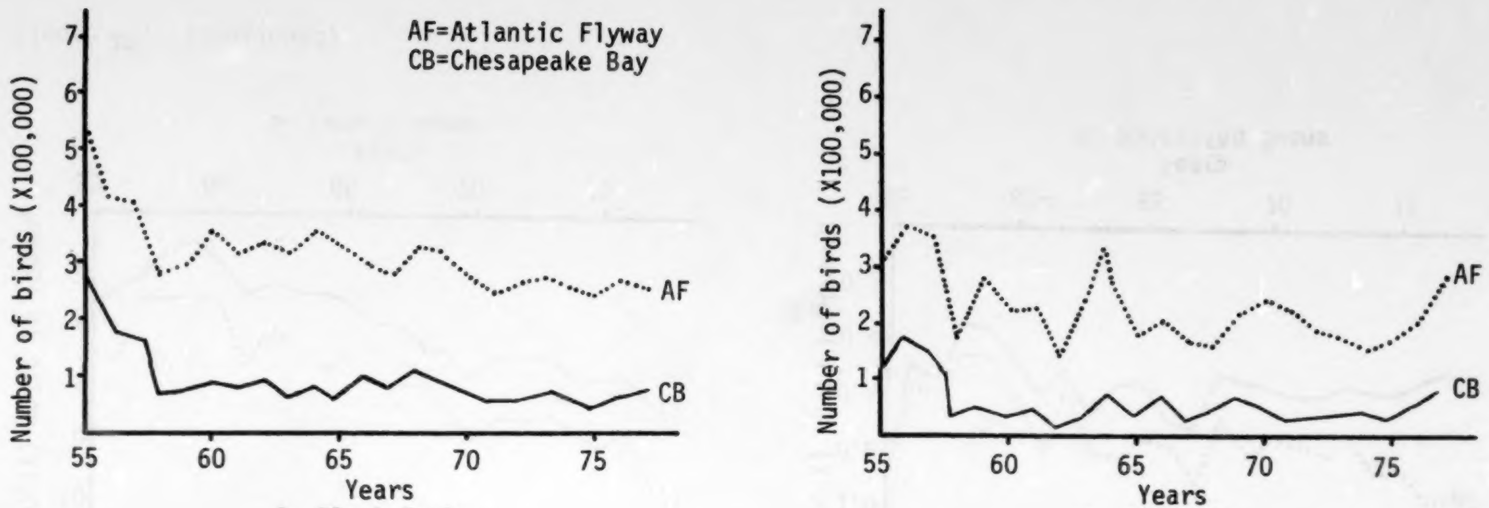
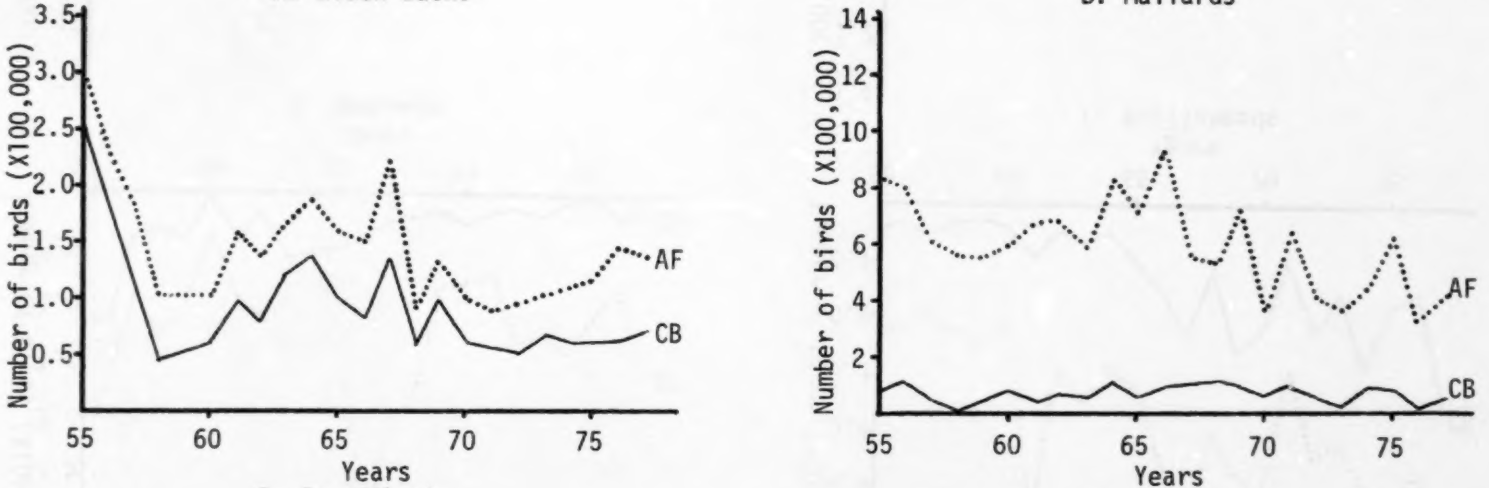
[a] Bellrose 1976



A. Black Ducks

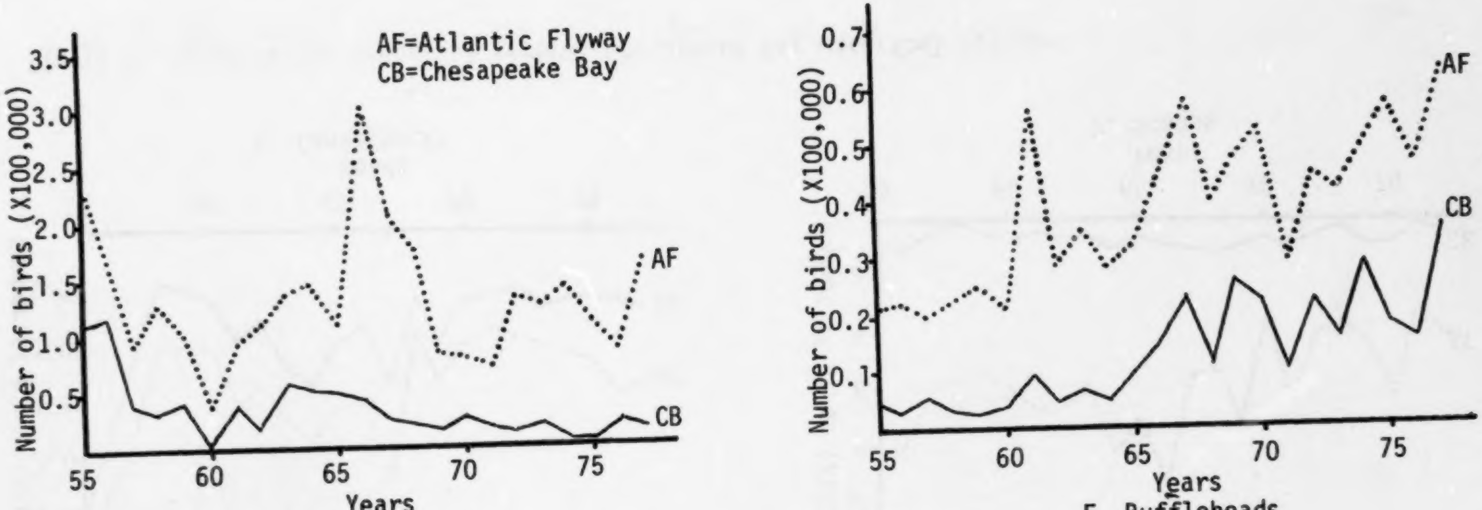
B. Mallards



C. Canvasbacks

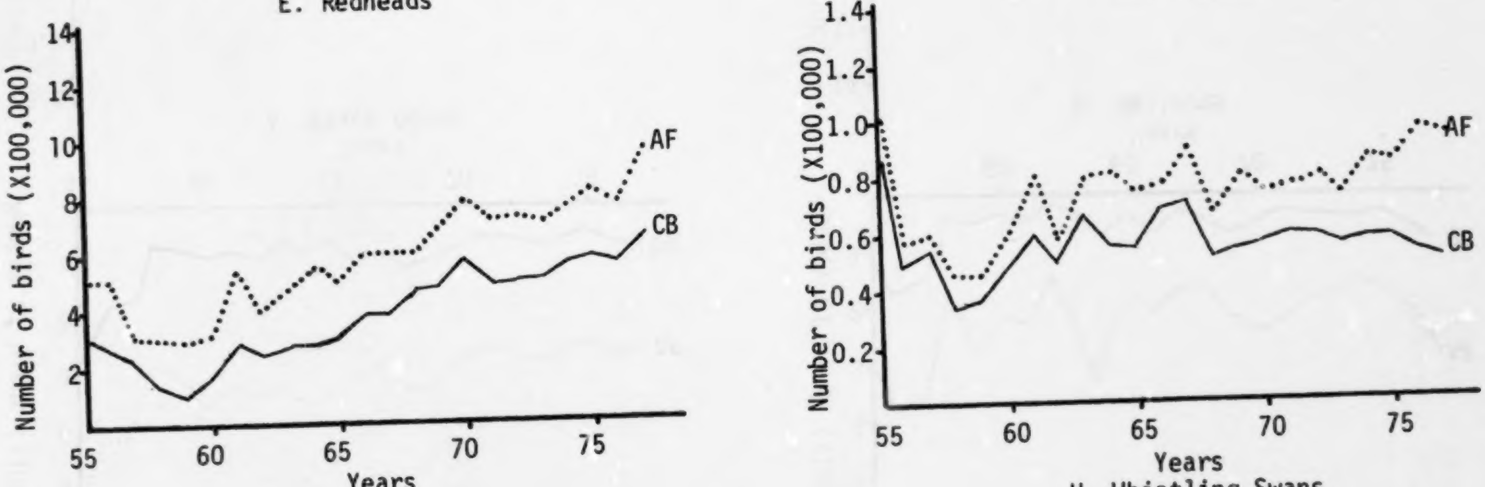
D. Scaups

Figure 32. Population trends of several Chesapeake Bay waterfowl species



E. Redheads

F. Buffleheads



G. Canada Geese

H. Whistling Swans

Figure 32. (Continued)

the late 1950s (Perry 1977). Data for individual puddle duck species show that population trends for the Atlantic Flyway parallel those of Chesapeake Bay (see Graphs A and B) though Mallards have undergone greater population fluxes in the Atlantic Flyway than the Chesapeake Bay.

Total populations for members of the diving ducks of genus Aythya showed trends similar to the puddle ducks (refer to Figure 32, C, D, E and F). The Canvasback populations of Chesapeake Bay in early survey years comprised nearly 50 percent of the Flyway population. The average total population size of Canvasbacks was higher in the 1960s when vegetation was more readily available in Chesapeake Bay. Canvasbacks have adapted to available invertebrates as their food resources owing to vegetation reductions.

Scaups, which are the second most important diving duck species in the Bay, remained relatively stable during the past 30 years in the Chesapeake Bay. The Bay populations of Redheads has been on the decline; Atlantic Flyway numbers increased erratically with a high in 1966 which was not reflected in the Chesapeake Bay population. This species shows the greatest dependence on SAV of all the diving ducks. In both the Atlantic Flyway and Chesapeake Bay, the population of the Buffleheads, a traditional animal feeder (McAtee 1911), has fluctuated upward.

The Canada Goose population has been rising steadily over the past 20 years (see Figure 32, Graph G). Field feeding on corn (Zea mays) has enlarged the range and carrying capacity for this species. Similarly, Whistling Swans have changed their feeding habits from submerged aquatic vegetation to field feeding (see Figure 32 Graph H). In the past decades, the population count of Whistling Swans has decreased slightly in the Bay and increased on the Atlantic Flyway.

The Mute Swan was recently introduced into the Eastern Shore of the Chesapeake Bay when five birds escaped from captivity in 1962. By 1970 there were approximately 27 birds inhabiting the Miles River area. Since 1970, the Mute Swan population has increased yearly by almost 40 percent (Fenwick personal communication). If this rate of increase continues, there could be about 10,000 mutes in the Bay by 1988. Bay area researchers are concerned that the Mute Swan may be causing interspecific competition for vegetable food resources of native waterfowl. A single Mute Swan can consume up to about 4.6 kg of SAV daily (Fenwick, personal communication).

Feeding Habits of Predominant Waterfowl of The Chesapeake Bay

Changes in the availability of traditional food sources is regarded by Bay area researchers as a primary cause of Chesapeake Bay waterfowl population fluctuations (Rawls, in press; Fenwick, personal communication; Stotts, personal communication; Perry et al. 1976). Not only have there been fluctuations in submerged aquatic vegetation abundance but also in the availability of the brackish water clam (Rangia cuneata).

In order to study past and present feeding habits, ducks can be conveniently divided into two groups based on physical morphology. Diving ducks exhibit a posterior leg position which makes terrestrial movement difficult and restricts feeding to an aquatic environment. Dabbling ducks have a more forward leg position which facilitates maneuverability on land but does not restrict aquatic movement. Table 17 lists in detail the feeding habits of the major waterfowl in the Chesapeake Bay. The following section discusses these feeding habits and the changes that have been noted.

Canvasbacks, one of the diving ducks, have fluctuated in number over the past few decades. Traditionally, wintering Canvasbacks were vegetation feeders, their mainstays being wild celery and sago pondweed. In the 1950s it was found that "Cans" existed on a mixed diet of vegetation and invertebrates (Stewart 1962). Recent declines of submerged aquatic vegetation has had an effect on the number of not only Canvasbacks but of most waterfowl wintering in the various regions of the Bay (Perry et al. 1976).

Canvasbacks have adapted their diet to abundances of particular foods. This flexibility has enabled the "Cans" to adapt to a situation of decreased submerged vegetation and become independent of vegetable foods. Currently their diet consists of about 48 percent animal and 52 percent vegetable matter (Rawls in press). Baltic and soft shelled clams constitute the main animal foods while corn and redhead grass are the primary vegetable foods. The availability of animals, particularly the brackish water clam may have caused yearly fluctuations in the Canvasback populations on the Western Shore. In 1968, only 11 percent of Canvasbacks wintered there; while in 1971 the population on the Western Shore increased to 87 percent. The availablity of Rangia to Canvasbacks has decreased due to diminished spawning and excessive clam size (Perry et al. 1976).

The Redhead, another diving duck, presently shows a greater dependence on vegetation than does the Canvasback. Gizzard studies by Rawls (in press) showed that the Redhead's diet consisted of about 76 percent vegetable and about 23 percent animal foods. Studies by Stewart (1962) showed that leaves, stems, root stalks and seeds of various species of rooted aquatics were the predominant foods.

The Lesser Scaup, also a diving duck, feeds on both vegetable and animal foods. Stewart (1962) showed that various species of mollusk were the principal food sources. Rawls (in press) determined animal foods comprised almost 48 percent of their diets. Percent vegetable intake over the 1959 to 1968 study constituted slightly over 52 percent.

Two other diving ducks, the Bufflehead and Goldeneye, are traditionally mollusk and crustacean feeders but will also consume vegetable matter (Stewart 1962). Rawls (in press) found that in the Chesapeake Bay, over 67 percent of the Buffleheads total food intake was animal. A 1962 study done in Back Bay and Currituck Sound showed that Bufflehead's subsisted on the seeds of widgeongrass and naiad, with plants comprising 64 percent of their diet (Sincock 1962). Recent work in Chesapeake Bay by Rawls (in press) showed that the Goldeneye has an animal diet similar to the Bufflehead: 63 percent animal food and 37 percent vegetable food.

Dabbling ducks such as Mallards and Black Ducks are able to feed in both aquatic and terrestrial environments. Mallards have been found to eat almost anything available (Bellrose 1976) but subsist almost entirely on a vegetable diet in the Chesapeake Bay (Stewart 1962; Rawls in press). The natural foods of Mallards differ sharply in freshwater estuarine marshes compared to brackish

Table 17. Food habits of waterfowl in the upper Chesapeake Bay, Maryland[a,b]

Waterfowl species	Animal food %	Vegetable food %	Total %	Predominant foods % total volume	
Canvasback	47.76	51.85	99.61	19.65	Baltic clam
				18.42	Corn
				16.32	Soft-shelled clam
				14.29	Redhead grass
				7.44	Widgeongrass
Redhead	23.40	76.59	99.99	29.29	Corn
				15.19	Redhead grass
				14.74	Widgeongrass
				10.53	Soft-shelled, Baltic and Mitchell's clams
				6.73	Conrad's false mussel
Lesser Scaup	47.56	52.47	100.03	20.48	Widgeongrass
				12.32	Soft-shelled clam
				11.59	Corn
				10.85	Redhead grass
				6.89	Mussel
Bufflehead	67.42	32.59	100.01	13.52	Widgeongrass
				11.85	Redhead grass
				10.00	Barnacle
				8.52	Fish
				7.22	Mud crabs
Goldeneye	63.09	36.87	99.96	19.44	Mud crab
				17.67	Corn
				14.88	Soft-shelled clam
				9.22	Barnacle
				9.00	Bivalves (unidentified fragments)
Mallard	5.00	94.80	99.80	24.14	Corn
				10.41	Redhead grass
				8.17	Widgeongrass
				9.13	Other submerged macrophytes
				1.64	Conrad's false mussel
				1.31	Soft-shelled clam
Black Duck	6.44	93.54	99.98	17.52	Corn
				15.50	Redhead grass
				14.20	Widgeongrass
				8.40	Milfoil
				1.91	Conrad's false mussel
				1.76	Amphipods
Canada Goose	0.00	100.00	100.00	32.42	Grasses (Gramineae)
				29.61	Corn
				6.97	Milfoil
				5.11	White clover
				2.99	Crab grass

---

[a] Based on waterfowl gizzards collected during 1959-1968 hunting seasons

[b] Rawls (in press)

water marshes within the Chesapeake Bay estuary. Stewart (1962) found that seeds of smartweeds, soft stem and three-square bulrushes and bur reeds predominated in freshwater, while in brackish water marshes, seeds of widgeongrass, pondweeds, smartweeds and the leaves and stems of submerged aquatics were more important food sources.

Black Ducks display feeding habits similar to Mallards with a diet consisting mainly of plant food (Stewart 1962; Rawls in press). One notable difference between Mallards and Black Ducks was a higher consumption of sea lettuce by Black Ducks (Rawls in press). Presently underway is a study by George Fenwick (The Johns Hopkins University) of about 300 duck gizzards collected in Eastern Bay, Miles River and Wye River from 1972 to 1976. Predominant species included in the analysis are Black Duck, Wigeon, Mallard,Scaup, Goldeneye, Bufflehead, Old Squaw and Ruddy Duck. Though the study has not as yet been completed, a tentative conclusion shows that submerged aquatic vegetation has been declining as a waterfowl food source (Fenwick, personal communication). Fenwick further hopes to show the impacts of the increased Mute Swan population on food availability to not only Whistling Swans but native diving and dabbling ducks.

Canada Geese are herbivores and have benefited from agricultural products more than any other waterfowl species of North America (Bellrose 1976). The population of this species has increased rapidly over the past three decades. It has been estimated that with modern crop harvesting methods, 10 percent of the crop remains after harvesting. Canada Geese have reduced their intake of aquatic plants and turned to croplands along the Atlantic Flyway.

Whistling Swans have a diet that is similar to Canada Geese. These swans now commonly feed on field grains and can cause crop damage (Sladen 1975). Early observations of Whistling Swans (Bent 1925) suggested that their diet was predominantly vegetable but they fed on some shellfish. Stomach analyses by Martin et al. (1951) indicated a diet of emergent and submerged vegetation with sago pondweed most common. Stewart and Manning (1958) found that Whistling Swans in brackish estuary waters consumed widgeongrass (50 percent) and mollusks (40 percent), in addition to small quantities of redhead grass and sago pondweed. In contrast, swans observed feeding in the fresh waters of the northern Chesapeake Bay fed entirely on wild celery.

The Role of Myriophyllum spicatum as a Waterfowl Food Source

The extravagant growth of milfoil that infested the Chesapeake Bay in the 1960s caused some concern among waterfowl enthusiasts as to milfoil's use as a waterfowl food source.

Based on information available at the time, Martin and Uhler (1939) had determined that milfoil was a less than satisfactory food for ducks. Florschutz (1969) analyzed the digestive tracts of three geese and 74 ducks from Back Bay and Currituck Sound and found milfoil in all of the geese. Wigeons had ingested large quantities of milfoil though Mallard and Black Duck digestive tracts showed little milfoil. The average food content for all 77 waterfowl showed that 45.5 percent was Eurasian watermilfoil. Gadwalls contained the highest (88.9 percent) by volume. Canada Geese followed with 83.0 percent which was higher than

American Wigeons (71.0 percent). Mallards and Black Ducks were lowest with 11.0 percent and 14.3 percent, respectively.

Florschutz (1969) indicated that although the availability of higher quality waterfowl food was reduced by the milfoil invasion, various waterfowl species used milfoil as a food source. Subsequently, the waterfowl populations of the inland waters of Virginia and North Carolina increased when total Atlantic Flyway populations decreased. Florschutz concluded that these population trends showed that milfoil was "not totally unacceptable" as a food resource.

Contrary to waterfowl trends of Back Bay and Currituck Sound in the 1960s, populations on the Susquehanna Flats showed marked declines. During this time, milfoil displaced more than a dozen native rooted aquatic species (Bayley et al. in press). January populations of waterfowl on the Flats averaged 4,900 during early infestation of milfoil (1958-1961). During the milfoil peak, January waterfowl counts on the Flats averaged only 390 birds. By 1965, when Vallisneria returned to about 50 percent the level sustained prior to milfoil invasion, an average of 4,860 birds wintered on the Flats.

Conclusions

An overall decline of Redheads and Whistling Swans suggests that the diminishing supply of a traditional food source of submerged macrophytes is a contributing factor. The Bufflehead, an animal feeder, has increased its population size in the Chesapeake Bay during the past few decades. Canada Geese, Mallards and Black Ducks have adapted to terrestrial feeding. Diving ducks such as Canvasbacks, have adapted to a more animal diet. Apparently, a decrease of a traditionally desired food source such as SAV results in several options for native and migratory waterfowl. They can either seek an alternative food source or compete for the diminishing food source. Either alternative could result in population reductions and locale changes.

AQUATIC FAUNAL ASSOCIATIONS

Submerged aquatic vegetation functions as an essential link in the aquatic faunal environment. SAV serves as both a direct and indirect food source; beds provide breeding and protection areas; and a variety of organisms use SAV roots, stems and leaves for attachment.

As a direct food source, SAV plant material, in both living and decaying forms, provides food for a variety of vertebrates and invertebrates. Physa eats the tender green shoots of Chara and Elodea; Planorbis lives entirely on submerged aquatic plants; Ancylus consumes decaying plant material; and Ruppia martima and Vallisneria spiralis can comprise up to 45 percent of the diet of the sheepshead (Archosargus probatocephalus). SAV, as an important source of detritus further serves as a food source for a wide variety of filter feeders.

By serving as habitat breeding and protection areas, SAV beds provide an indirect food source for many species of fauna. Plant roots, stems and leaves provide firm bases for attachment of such sedentary epibenthos as mussels, barnacles and mollusks (Green 1968). Fish, such as the cownose ray (Rhinoptera bonasus) actively feed on mollusks that inhabit SAV beds (Orth 1975a). Marsh

(1973) observed the predation by numerous fish on the epifauna of the aquatic plants. *Minidia minidia* (common silversides), *Apeltes quadracus* (four-spined stickleback) and *Syngnathus fuscus* (pipefish) were the most abundant fish observed.

Baker (1918) found associations of *Potamogeton perfoliatus*, *Vallisneria spiralis*, *Elodea canadensis*, *Ceratophyllum demersum*, *Myriophyllum spicatum* and *Chara* spp. with the mollusks *Acella haldemani* living on the leaves and stems and in the vegetation. *Bythinia tentaculata*, *Galba cataseopium*, *Planorbis* spp., *Physa ancillaria* and the crustacean *Gammarus fasciatus* were among the other inhabitants. For a 450 sq m area, Baker estimated the animal life on *Potamogeton* sp. and *Myriophyllum* sp. as:

Potamogeton sp.	*Myriophyllum* sp.
247,500 mollusks	45,000 mollusks
90,000 associated animals	56,250 associated animals
337,500 Total fauna	101,250 Total fauna

These values were calculated from fresh water areas, but salt water areas characteristically have larger population numbers.

Numerous insects and insect larvae use the SAV for food and attachment. Damage can be done to aquatic plants by such insects as various leaf beetles (*Chrysomelidae*), snout beetles (*Curulionidae*), caddisfly larvae (*Leptoceridae* and *Hydroptilidae*) and froghoppers (*Delphacidae*) (Martin and Uhler 1939).

As a consequence of the wasting disease of *Zostera marina* in 1930 and 1931, the population of small motile seed of the scallop (*Pecten*) *irradians*) severely declined and all but destroyed the commercial scallop activity (Linduska 1964; Cottam and Addy 1947). Radical adjustments were necessary for the spat of other shellfish which depended on eelgrass for a place of anchorage. Shellfish such as crabs, scuds and other crustaceans were also sheltered by these dense beds. The eelgrass also provided protection in its dense foliage for the young of many species of fish (Cottam and Addy 1947).

The blue crab (*Callinectes sapidus*) is known to spend much of its maturation period dependent on SAV beds for sanction (Darnell 1959). Once caught and prepared for shipment, soft-shelled crabs are often packed in eelgrass and crushed ice (Anonymous 1959). Fish, for example the black bass, sunfish and bluegill, are known to build nests out of pondweed when breeding in shallow waters. Still others, Cyprinus carpio and Aineirus nebulosus seek marshy or swampy areas characterized by dense SAV (Baker 1916).

EPIPHYTE ASSOCIATIONS

Aquatic macrophytes growing in natural conditions typically support a dense epiphytic community. Historically, only the number, species composition and biomass of the epiphytes have been examined intensively (Krecker 1939; Skerman 1956; Hargraves 1965; Marsh 1973). In more recent years there has been an increased interest in interactions between the host and its associated epibiota. Prouse (1959) first indicated the possibility of such interactions by showing statistically significant differences between epiphyte populations on three different species in the same environment. With the advent of sensitive chemical techniques and the initial delineation of the role of dissolved organic matter in aqueous environments, it has become increasingly clear that the epiphyte-macrophyte association is complex and the interactions numerous.

The epiphytic community of macrophytes in a littoral zone may have a biomass of significant size and, in an area heavily grown over with submerged aquatic vegetation, may have a biomass greater than the phytoplankton in the immediate area (Allen 1971). Primary productivity of the epiphytic flora in some areas may be among the highest recorded, especially if the macrophytic vegetation is well developed (Allen 1971; Wetzel 1964). Furthermore, Allen showed that the bacterial populations on the leaves of Chara and Nitella had high activity rates in the utilization of glucose and acetate. That this high rate of productivity and chemo-organotrophy exists, even in areas of low ambient nutrient levels, suggests that the epiphytic flora is possibly being subsidized to a certain extent by the host, aside from a locally enriched oxygen level.

Macrophytic algae and phytoplankton excrete dissolved organic carbon under many conditions (Hellebust 1965; Fogg 1966; Khailov and Burlakova 1969), although the extent to which this occurs has been questioned recently by Sharp (1977). Khailov and Burlakova (1969) suggested that dissolved organic material represents another port of entry into the food web, in addition to the traditional grazing and detrital pathways.

Excretion by aquatic angiosperms has also been noted. Najas flexilis, grown under sterile conditions, was found to excrete significant proportions of total carbon fixed as dissolved organic matter (Wetzel 1969; Allen 1971). Zostera marina excreted organic matter at the rate of 0.6 to 2.5 percent of total carbon fixed (Penhale 1977). It also appears likely that with the onset of senescence, organic output is increased. The numbers of bacteria epiphytic on Elodea, that are capable of utilizing glucose were found to increase nine-fold as the leaves became senescent (Ramsey, 1974). Epiphytic flora and fauna, due to spatial configurations with the host plant, are ideally positioned to use organic matter and nutrients liberated by the host. Many species of animals and plants capable of an epiphytic existence can make use of dissolved organic matter (Stephens 1967; Provasoli 1971).

It now appears likely that the excretion of dissolved organic matter is responsible, in part, for the high rates of physiological activity noted earlier. Several studies have shown that transfer of material does exist between host and epiphyte. Harlin (1973) showed reversable, light-independent, translocation of photosynthate between *Zostera marina* and an epiphytic Rhodophyta, *Smithora naradum*. Isotopes of carbon and nitrogen taken up in the rhizosphere of *Z. marina* were found to be transferred to the natural epiphytic flora within one hour (McRoy and Goering 1974). In a more indirect study by Penhale (1977), the release of dissolved carbon by *Z. marina* was monitored in three experimental situations: (a) *Z. marina* alone; (b) *Z. marina* plus epiphytes; and (c) epiphytes alone. She reported significant differences in the percentage of released carbon to carbon fixed between situation (a) and (b), and between (b) and (c). Differences between "clean" plants and epiphytes alone were not significant. The conclusion was reached that this difference was due to an interchange between the host and epiphytes involving transfer of organic material.

Fry and Ramsey (1977) and Ramsey and Fry (1976) treated two species of aquatic plants, *Elodea canadensis* and *Chara vulgaris*, with the herbicide paraquat to observe changes in the activity rates of the attached microbial flora. *E. canadensis*, which is sensitive to the herbicide, showed an increase in heterotrophic activity of its epiphytes, while *C. vulgaris*, a resistant species did not. This was accounted for by the increase in extracellular matter given off by the dying *E. canadensis*. Although both plants were in the same immediate vicinity, the epiphytes of *Chara* did not respond to the pulse of nutrient and dissolved matter given off by *Elodea* indicating that the host plant probably provides a substantive portion of the substrate for the epibiota.

Excreted carbon may also affect many animal species as well, both directly and indirectly. For example, algal exudates were found to influence the selection of substrata by the marine peritrich ciliate *Vorticella marine* (Langlois 1975). This may be due to direct chemical influence or indirect enhancement of growth of the bacteria on which *Vorticella* feeds. Although elucidated in the algae, there is no reason to suspect a different case in the angiosperms.

Extracellular products produced by macrophytes probably do not all increase activity of the epiphytes. Sieburth (1968) lists a variety of antibiotic agents released by macrophytic algae. Some of these, including tannins and other polyphenols, may be excreted by submerged aquatics and interact with the epiphytic community (Harlin 1973). Macrophytes in the field have been noticed to produce an inhibitory effect on epiphytes in a freshwater lake (Fitzgerald 1969). The effect is thought to be a nutritional competition (epiphytic growth was abundant only when nitrogen levels were high) or a toxic principal released by the macrophytes (Fitzgerald 1969). *Ceratophyllum demersum* was found to inhibit the growth of several cyanobacteria of the order Nostocales when grown together in non-limiting media; however, a dilute water extract of the angiosperm had a stimulating effect on the same organism (Kogan and Chinnova 1972).

Release of dissolved organic matter by higher plants represents an inefficiency that, at first, would seem to be non-adaptive. If this excretion enhances the growth of epiphytes, it would lead to a reduction in light to the photosynthetic units of the higher plant and a decrease in bicarbonate flux as pointed out by Sand-Jenson (1977). It would seem that this release would be selected against unless a feedback mechanism existed to explain this apparent paradox. Several mechanisms have been suggested. An encrusting layer of epiphytes on leaf surfaces may serve a protective function against biting and sucking herbivores (Hutchinson 1975; Sieburth and Thomas 1973). Another possibility is the passing of nitrogenous compounds to the higher plants by nitrogen-fixing epiphytes. Goering and Parker (1972) presented evidence that nitrogen-fixing epiphytes of sea grasses in Redfish Bay, Texas, contributed to the nitrogen budget of the sea grass community. Patriquin and Knowles (1972) found high levels of fixation in the rhizosphere of tropical sea grasses, but less so in the temperate Zostera marina. However, other investigators (McRoy et al. 1973), found no measurable nitrogen fixation in the communities associated with Z. marina and concluded that this activity might be more common in tropical environs where ambient levels of nitrogen might be less. Capone and Taylor (1977) working with the epiphytic community of Thallisera testudimum found that nitrogen fixation (mainly of the genus Callothrix) may provide 8 to 38 percent of the daily nitrogen requirement for leaf production. That nitrogen fixing activities may be enhanced by the extracellular production of reduced carbon substrates (Head and Carpenter 1975) lends support to the possibility of a nitrogen-fixing feedback mechanism.

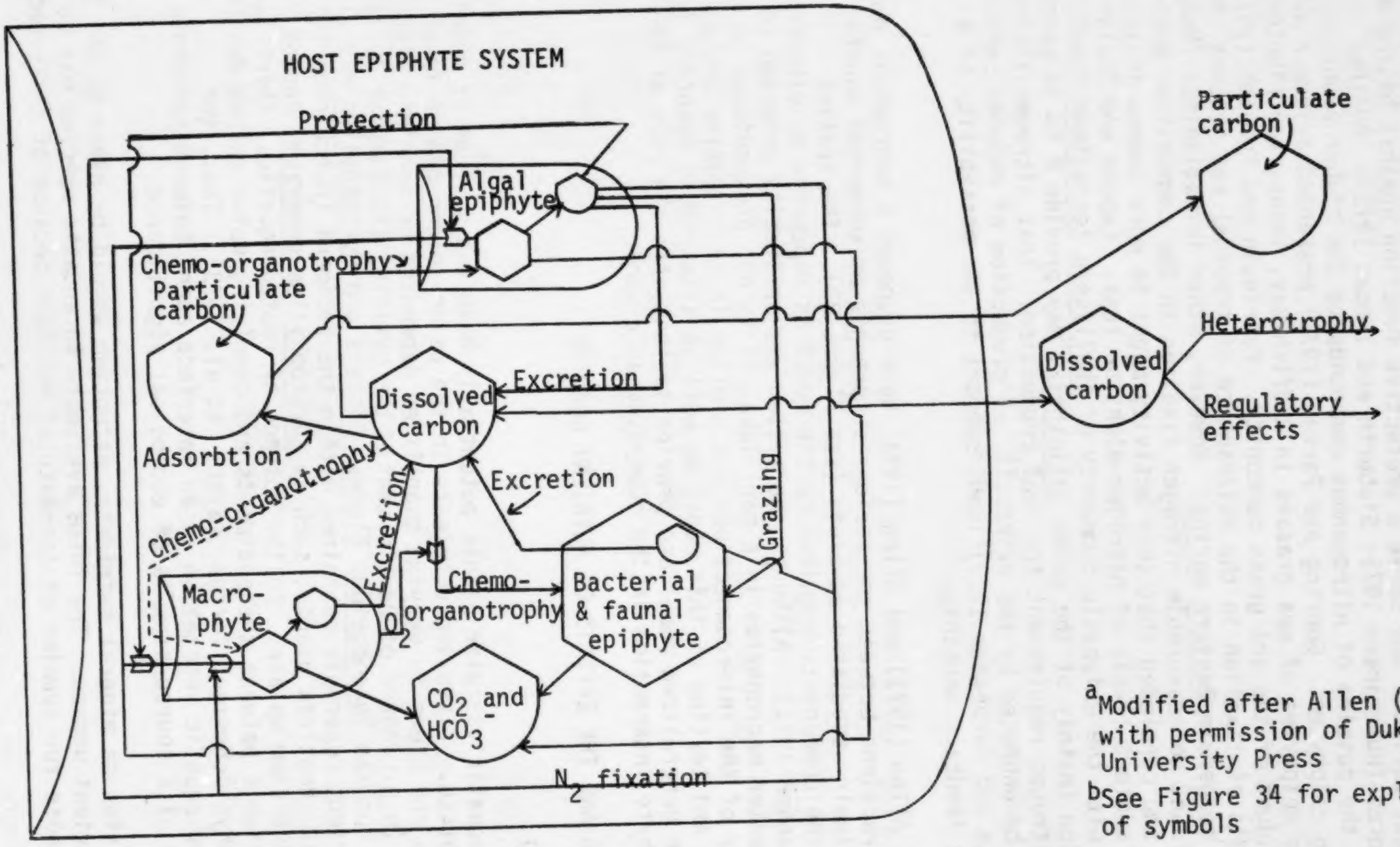
Wetzel and Allen (1971) and Allen (1971) have proposed a conceptual model to explain interactions between the various components of submerged aquatic vegetation and their associated epibiota (see Figure 33). The spatial arrangement of the components required by the model is supported by electron microscopy (Allanson 1973). Although the system was originally proposed to deal with freshwater macrophytes in a marl lake, it is not presumptuous to assume that many of the interconnections are similar in an estuarine system. The application and testing of this model as well as attempts to quantify it, may yield a more generalized model and provide insight into the role of the macrophyte-epiphyte interaction in the ecosystem as a whole.

NUTRIENT CYCLING AND THE EFFECTS OF NUTRIENT LOADING

Nutrient cycling

Submerged aquatic vascular plants potentially have two sources of nutrients available for uptake. Dissolved nutrients in the water column can be taken up by the leaves and stems in varying quantities depending on specific plant characteristics (e.g. degree of development of the cuticle) (Sculthorpe 1967). Rooted aquatics such as Potamogeton, Elodea, Zostera and Myriophyllum may also draw on the nutrient reservoir contained within the sediment (Hutchinson 1975, Sculthorpe 1967). Rootless plants, such as Ceratophyllum demersum, depend solely on leaf and stem uptake. In the absence of rooted aquatics, there is an exchange of nutrients between the sediments and overlying water column due to biological and physiochemical forces (Syers et al. 1973). Thus, the role of submerged aquatic vegetation as an interface agent between sediments and pelagic zone and as a source/sink is of ecological significance.

In a discussion on mineral nutrients, attention should be given to the form in which the nutrient occurs, The forms available to aquatic species may be different than those for species of terrestrial habitats because of differences


HOST EPIPHYTE SYSTEM
Protection
Algal epiphyte
Chemo-organotrophy
Particulate carbon
Dissolved carbon
Excretion
Adsorbtion
Chemo-organotrophy
Excretion
Excretion
Grazing
Macro-phyte
O_2
Chemo-organotrophy
Bacterial & faunal epiphyte
CO_2 and HCO_3^-
N_2 fixation
Particulate carbon
Dissolved carbon
Heterotrophy
Regulatory effects

[a]Modified after Allen (1971) with permission of Duke University Press

[b]See Figure 34 for explanation of symbols

Figure 33. Conceptual model of host-epiphyte interaction[a,b]

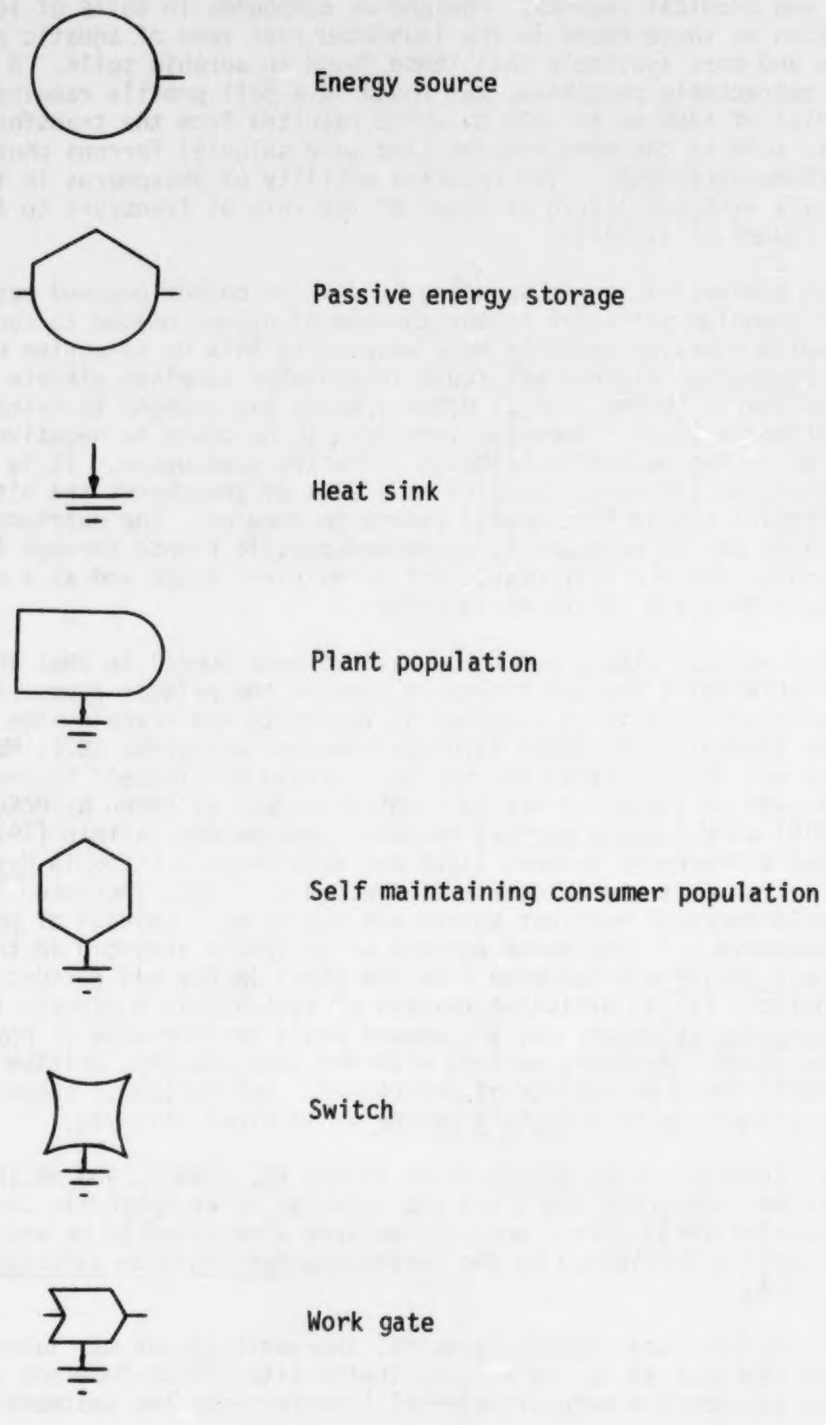

Energy source
Passive energy storage
Heat sink
Plant population
Self maintaining consumer population
Switch
Work gate

Figure 34. Energy language symbols (Odum 1972)

in physical and chemical regimes. Phosphorus compounds in soils of low redox potential, such as those found in the inundated root zone of aquatic plants, are more soluble and more available than those found in aerobic soils. A marked increase in extractable phosphorus was found in a soil profile ranging from a redox potential of +200 mv to -200 mv which resulted from the transformation of the ferric salt to the more reduced (and more soluble) ferrous phosphate (Patrick and Mahapatra 1968). The relative mobility of phosphorus is increased in anoxic soils although little is known of the rate of transport to the overlying water (Syers et al. 1973).

Nitrogen mineralization in reducing conditions cannot proceed past the formation of ammonium salts due to the absence of oxygen needed to convert to nitrate. Rooted aquatics probably have adapted to this by selective uptake of ammonium. *Potamogeton alpinus* was found to die when supplied nitrate as the sole nitrogen source (Misra 1938). Other species may respond to nitrate, however (Sculthorpe 1967). Ammonium ions tend to be bound to negatively charged clay particles in the sediments although diffusion does occur. It is thought, therefore, that the sediments function as a sink of phosphorus and nitrogen and as a potential source for aquatic plants to draw on. The nutrient pool in the water column can be accessed by submerged aquatic plants through the leaves and stems. Submerged vascular plants act as nutrient traps and as a sink for these dissolved minerals (Clark et al. 1973).

Submerged aquatic plants may serve as "nutrient pumps" in that the roots can trap available nutrients and transport them to the pelagic zone. Inorganic phosphate has been shown to be absorbed by the roots and translocated throughout the plant (McRoy and Barsdate 1970; Bristow and Whitcombe 1971; McRoy et al. 1972; DeMarte and Hartman 1974) and may be excreted or "leaked" through the leaves. The rate of transport may be light-dependent as shown by McRoy and Barsdate (1970) with *Zostera marina*; however, DeMarte and Hartman (1974) found no significant differences between light and dark translocation in *Myriophyllum spicatum*. If transport and release are affected by light, increased turbidity and depth would decrease nutrient uptake and excretion. Leakage of phosphorus may be considerable. Thirty-three percent of phosphate absorbed in the root zone of *Zostera marina* was excreted from the plant (McRoy and Barsdate 1970). DeMart and Hartman (1974) exhibited release of radioactive phosphate from intact *Myriophyllum spicatum*, and the amount could be increased by physical injury to the plant. However, working with the same species, Bristow and Whitcombe (1971) found no leakage of phosphorus. Harlin (1973) showed a transfer of inorganic phosphorus from *Zostera marina* to an algal epiphyte.

Nitrogen supplied to *Z. marina* roots in the NO_3^-, NH_4^+, and $NH_2(CO)$ forms was translocated throughout the plant and excreted to an epiphytic community (McRoy and Goering 1974). Iron and calcium were also found to be absorbed from the sediment and released by the macrophyte *Myriophyllum spicatum* (DeMarte and Hartman 1974).

In several freshwater aquatic species, inorganic carbon was taken up by the roots and transported to the photosynthetic site (Wium-Anderson 1971). This may also represent a mode of material transfer from the sediments.

Rooted aquatic vascular plants are therefore capable of withdrawing mineral nutrients from the anoxic zone and pumping them upward. Nutrients can also be trapped from the water column and incorporated into biomass. Release of these nutrients can occur by excretion by healthy plants or by the death and subsequent decay of individuals.

Responses to Nutrient Loading

An increase of allochthonous nutrients into the aqueous system may produce several effects on submerged vascular plants. With an increase of ambient nutrient levels, levels within the plant will increase as well and may be biologically concentrated (Anderson et al 1967; Allenby 1968). This may prove to be a useful tool in the assessment of nutrient loading in that changes in the chemical analysis of plant tissue would reflect changes in exterior levels of nutrients (Sculthorpe 1967; Adams et al 1973). Changes in vegetative morphology may occur as suggested by Adams et al. (1971) for Elodea. Increases in nutrient levels may also increase productivity in submerged species to a certain point and several species have been shown to exhibit "luxury consumption" (Orth, in press; Ryan 1969).

Phytosociological patterns may develop in response to nutrient levels. Early workers (Pond 1905; Pearsall 1920; Misra 1938; Swindale and Curtis 1957) found that the distribution of aquatic macrophytes was dependent on the chemical nature of the substrate. Jaworski et al. (1972) using historical observations of vegetation patterns in the Potomac River estuary, correlated increases in nutrients with a succession of species. In their example, water chestnut (Trapa natans) was replaced by Myriophyllum spicatum which was in turn replaced by a cyanobacterial bloom of Anacystis. This succession was due to increases in nitrogen and phosphorus and little correlation was found with changes in available carbon. The same successional pattern was observed in rivers of the upper Chesapeake Bay in response to nutrient loading (Clark et al. 1973). An increase in phytoplankton in response to nutrient loading, as seen above, may limit light and decrease productivities of macrophyte communities.

Predictions of the response of submerged vascular plants to nutrient loading based on present knowledge would be equivocal. More research is necessary if answers to these questions are to ascertained.

SUBMERGED AQUATIC MACROPHYTES AS BIO-INDICATORS

Submerged aquatic species are not commonly viewed as indicators of biological conditions or trends within the Chesapeake Bay. The aquatic grasses do not respond as rapidly to environmental conditions as various algal, phytoplankton and zooplankton species that are capable of rapid growth in bloom proportions.

Sculthorpe (1967) suggested that SAV species are potentially useful as indicators of pollution trends. Adams et al. (1971) supported Sculthorpe's suggestion with evidence as to chemical and morphological responses of Elodea canadensis to increasing levels of nutrient pollution. As of their writing, Adams et al. indicated that no experimentally comprehensive work had been performed in order to evaluate the potential advantages or disadvantages relating to the use of SAV as aquatic pollution trend indicators. They further suggested the need for clarification as to the identification of specific nutrient pollution conditions in relation to individual SAV species.

In assessing the value of using plants as indicators in place of modern water pollution analysis technology, Adams et al. (1971) pointed out that relative costs, scope and manpower requirements should be compared. They concluded that automated monitoring systems are initially costly, expensive to maintain and limited in scope. By comparison, water monitoring through the use of pollution-sensitive SAV would not only reduce initial and maintenance costs but could provide a wide coverage using essentially a single instrument. Adams et al. suggested a system of transporting trays of stock plants to various locations and the selective breeding of species sensitive to specific pollutants.

The floating aquatic duckweed (Lemna perpusilla) has been used by Feder and Sullivan (1969, abstracted in Thomas et al. 1973) in experiments with ozone. Plants treated with a low concentration of ozone over two weeks were slower to multiply, had a lower rate of frond doubling and were slower to produce fewer flowers than control plants.

Several species of submerged and floating macrophytes have been tested for effects from smelter pollution in Ontario (Gorham and Gordon 1963 , abstracted in Thomas et al. 1973). Species occurrence was found to be inversely related to dissolved sulphate. Species diversity was low even where sulphuric acid was almost totally neutralized and waters were above pH 6. Utricularia vulgaris and Potamogeton epihydrus v. nuttallii appeared to be rather sensitive to heavy metal concentration.

Burrows (1971, abstracted in Thomas et al. 1973) indicates that Ulva lactuca has potential as a pollution indicator species because of its ease of culture and sewage pollution reactions. Laminaria saccharina is also listed as a possible indicator species due to its sensitivity and availability. An historic view of eutrophication in the Potomac River estuary (Jaworski et al.1972) suggests that nuisance plant conditions did not develop linearly with an increase in nutrients. As nutrient loading increased in the estuary, nuisance plant growth was favored based on individual species nutrient requirements. Thus, Trapa natans infested the Potomac in the 1920s to be replaced by Myriophyllum spicatum in the 1950s and 1960s. As milfoil died out in the mid 1960s. it was replaced by the blue-green alga Anacystis. The massive blooms of Anacystis that have occurred each summer since 1960 can be associated with phosphorus and nitrogen loading increases.

TURBIDITY AND WATER MOVEMENT

Submerged aquatic plants tend to modify their physical environment through their growth habits. By providing resistance to stream flow and wave action, hydrophytes not only reduce average water movement but aid in the settling out of silt particles thus reducing turbidity and building up substrates. (Sculthorpe 1967) Macrophytes initially provide frictional resistance to current velocity.

Some species of macrophytes, ideally those SAV with linear ribbon-like leaves do not provide finely dissected surfaces that trap silt particles and organisms. Thus particulate matter settles, becomes established and provides bottom sediments that are often conducive to the establishment of further SAV. Further colonization results in yet slower currents and greater sediment deposition (Mitchell 1974). Sculthorpe (1967) cited Millebrand (1950) as having found a 75 percent reduction in maximum current velocity in SAV colonized river areas.

SEDIMENT STABILIZATION

High energy shorelines are not conducive to colonization by SAV. However, in areas where vegetation is established, grass beds aid in the deposition and stabilization of bottom sediments. As plants provide resistance to currents, particulate matter settles and builds up a substrate of fine silt particles that enhances plant colonization. As plant root stock developes and the SAV beds expand, bottom substrates are further stabilized.

pH

Hydrogen ion concentration, or pH in an aquatic environment, is a function of the dissolved CO_2 content: as the CO_2 content is lowered, pH increases. Thus as plants photosynthesize during the day and lower the dissolved CO_2, the pH will increase. At night as plants respire and give off CO_2, the pH will decrease.

The degree of change in pH is greatly determined by the buffering capacity of the aquatic medium. Carbon dioxide combines with water to form H_2CO_3 which in turn reacts with limestone to form carbonates ($-CO_3$) and bicarbonates (HCO_3). This carbon dioxide-carbonate-bicarbonate complex acts as a buffer and results in more neutral conditions.

Aquatic plants thus affect pH through CO_2 fluctuations and pH in turn affects plant chemical processes. Enzyme activity rate has been determined to be influenced by the pH of the medium as enzymes exhibit optimum activity within specific pH ranges (Small 1946). pH further influences imbibition or swelling of proteins and is thus related to seed germination. Hydrogen ion concentration may alter heat susceptibility and enzyme solubility and is related to absorption and accumulation of salts (Curtis and Clark 1950). Toxicity and solubility of heavy metals, detergents, aromatic solvents, acids, alkalis and salts may also be influenced by pH (Sculthorpe 1967).

COMMERCIAL AND CONSERVATION VALUES

In various parts of the world, species of submerged aquatic vegetation are considered a resource and used for forage and fertilizer. Commission on International Relations (1976) includes such examples as the use of water hyacinth (*Eichornia* spp.) for bedding material in mushroom cultivation in the Philippines, as a substitute for German peat moss, as a propagation medium for houseplants and as a nutritional supplement for nonruminant animals in Southeast Asia.

Boyd (1974) discussed the food value of several emergent and submergent aquatic plants including Najas guadalupensis, Ceratophyllum demersum and Potamogeton diversifolium. Table 18 compares these three SAV with alfalfa.

Table 18. Food value (percent dry weight) of several species of submerged aquatic vegetation and alfalfa[a]

Species	Ash %	Crude Protein %	Fat	Cellulose %
Najas guadalupensis	18.7	22.8	3.8	35.6
Ceratophyllum demersum	20.6	21.7	6.0	27.9
Potamogeton diversifolius	22.7	17.3	2.8	30.9
Eichornia crassipes	18.0	17.1	3.6	28.2
Alfalfa	8.6	18.6	2.6	23.7

[a] Boyd 1974

Despite the high nutritive levels in some SAV species, they also have a high water content. In order to best use the plant material for fodder, the 90 to 95 percent water content should be lowered by dehydration in order to increase the food value per weight. This need for drying increases the expense and decreases the practicality of using SAV for fodder (Boyd 1974).

In the United States, the National Aeronautics and Space Administration (NASA) is conducting research on the use of water hyacinth and other aquatic plants as a source of methane gas. Also being investigated is the use of such SAV species as Elodea canadensis and Ceratophyllum demersum in water-treatment systems (Commission on International Relations 1976). Water hyacinth is already being used in Lucedale, Mississippi, by a NASA facility to remove nutrients from sewage effluent (Commission on International Relations 1976). The harvested plant material is then processed for animal feed, fuel and fertilizer. Duckweed (Spirodela sp.) is being grown at Louisiana State University on dairy farm effluents then substituted for alfalfa for swine and dairy feed (Commission on International Relations 1976).

Native submerged aquatic vegetation in the Chesapeake Bay is generally regarded more for its beneficial use by wildlife than its possible commercial use. Field work is presently being conducted by Vernon Stotts, Maryland Wildlife Administration to determine the feasibility and success of planting methods for Ruppia maritima. Further work on this variety would be necessary in order to establish the usefulness of SAV in the establishment of conservation areas, the possible use of SAV as bio-indicators and the feasibility of commercial SAV uses.

CHAPTER 3

HISTORICAL TRENDS OF CHESAPEAKE BAY SUBMERGED AQUATICS

INTRODUCTION

Historic documentation of the distribution and abundance of aquatic vegetation in the Chesapeake Bay is sparse until the late 1950s. Up to that time, documentation is limited primarily to occasional newspaper references, herbarium notations, personal communications and occasional scientific papers dealing with single-season populations found in specific areas. The oldest records available are found in the Smithsonian's herbarium dating back to 1871. From the late 1920s through the present, Francis Uhler of the U.S. Fish and Wildlife Service Migratory Bird and Habitat Research Laboratory (MBHRL), has observed and noted Bay grasses, especially in the Potomac River.

The first comprehensive survey of SAV was begun in 1958 on the Susquehanna Flats by MBHRL personnel. Located in the upper Bay at the mouth of the Susquehanna River, this area has always been important for waterfowl feeding. This study is presently being continued and documents the occurrence and relative abundance of all the major species of SAV on the Flats.

In 1959, Vernon Stotts of the Maryland Wildlife Administration designed a transect system for sampling Maryland Chesapeake Bay benthos in the autumn (Maryland Pittman-Robertson Project W-30-R-8). Based on nine transect lines, this Benthic Survey was continued through 1961.

In 1967, Vernon Stotts again designed a summer vegetation sampling program (Maryland Pittman-Robertson Project W-45-2). This survey dealt only with SAV and was based on an intensive network of transects radiating from the shorelines. The Vegetation Survey covered the Maryland Eastern Shore and was continued through 1969.

The idea of this survey was picked up by James A. Kerwin and Robert E. Munro of the MBHRL and summer sampling began in 1971 with the aid of Vernon Stotts. This survey is presently in operation and is the only Maryland Bay-wide program in existence for rooted aquatics.

Available information for SAV in Virginia waters is limited mainly to herbarium specimens and the work of Robert Orth and Walter Priest from Virginia Institute of Marine Science. Orth has studied extensively the history of Zostera marina in the Bay as well as searching literature on the world-wide distribution and abundance of eelgrass. Priest surveyed the Rappahannock and Piankatank Rivers in the spring of 1977. Preliminary results of the 1977 MBHRL Survey of the Potomac River have been included in the conclusion section of this chapter. these data are for both Maryland and Virginia.

These surveys constitute the main body of available knowledge on the distribution of SAV in the Bay. Along with site-specific studies, herbaria specimens, newspaper accounts and personal communications, the findings of the surveys will be presented later in this chapter.

CHESAPEAKE BAY CHRONOLOGY

In attempting to assess an historic documentation of declines in submerged aquatic vegetation in the Chesapeake Bay, it is first necessary to view the situation chronologically. Table 19 presents almost fifty years of Bay area history including major storms,droughts, population growth, pest plant infestations and abnormal precipitation levels. The following section discusses many of these major events and their immediate results.

1920s

Water chestnut (Trapa natans) first appeared near the mouth of Oxon Run at the head of the Potomac River in 1923 (Rawls 1964). Within a decade, the species had spread over forty miles of the river in shallow water areas. Waterfowl breeding grounds were termed useless (Maryland Department of Tidewater Fisheries 1955) presumably because milfoil is generally considered to be one of the least desirable waterfowl foods.

1930s

In the early 1930s, Francis Uhler, MBHRL is reported to having seen ducks feeding on redhead grass and wild celery from the old 14th Street Bridge in Washington, D. C. (Rawls et al. 1975).

On August 23, 1933, the Bay was hit by the century's worst storm. Water rose about 45 cm per hour sending 0.6 to 0.9 m high waves over land and resulting in 6 to 9 m high waves in the ocean. The water level was almost three feet higher than the "great centennial" storm of 1876 (Daily Banner), August 25, 1933).

On September 16, 1933, another storm hit, raising water levels to similar heights as the storm of the previous month. The Daily Banner reported September 20, 1933, that thousands of tons of eelgrass and peat plants were destroyed in the Nanticoke, Blackwater and Hooper Island area by excessive wave action.

From 1930 to 1932, an unprecedented drought brought brackish water farther up into the estuaries (Stewart 1962). This impact, coupled with the crowding out effects of an increasing abundance of Trapa natans, caused SAV to gradually disappear by 1935 (Gwathmey 1945; Stewart 1962). The impact of the drought followed by the 1933 storms contributed to about five years of severe biological stress to the Chesapeake Bay ecosystem.

Table 19. Chesapeake Bay chronology

1930	1930's _Trapa natans_ infestation in Upper Potomac River; 1930's _Zostera_ decline
1931	Severe drought
1932	Hurricane (August) -- extreme tides
1934	Residual severe biological stress from 1933
1935	
1936	Record flood (March)
1937	
1938	
1939	
1940	Bay area population 1,600,000
1941	
1942	Flood (October)
1943	
1944	
1945	
1946	
1947	
1948	
1949	
1950	Approximately 4,000 ha of Potomac infested with _Trapa natans_
1951	
1952	
1953	
1954	Hurrican Hazel (October)
1955	Tropical storm Connie (August); tropical storm Diane (August)
1956	Northeastern (April)
1957	One-third 1958 rainfall (May to August)
1958	
1959	
1960	Tropical storm Donna (September); Bay area population 2,600,000
1961	Approximately 20,000 ha of Bay covered with _Myriophyllum_
1962	Northeastern (March); over 40,000 ha of Bay covered with _Myriophyllum_
1963	Drought from 1963 to 1965
1964	Salt water intrusion from 1964 to 1966; drought
1965	Drought
1966	
1967	Excessive precipitation (May and June)
1968	
1969	Abnormally low precipitation (winter); high precipitation (summer)
1970	Bay area population 3,100,000
1971	
1972	Tropical storm Agnes (June)
1973	
1974	
1975	
1976	Exceptionally low precipitation
1977	Record cold winter

By 1933, 4,000 ha of *Trapa natans* thrived in the Potamac River from Quantico north to the falls. Satisfactory control was achieved through underwater mowing techniques by the U.S. Army Corps of Engineers (Rawls 1964).

Along the east coast of North America and the west coast of Europe, major declines in *Zostera* occurred in the 1930s. The causal organism may have been the protozoan, *Labyrinthula.* Discussions as to the decline of eelgrass are presented in Chapter 1 in the section dealing with *Zostera marina* and in Chapter 4 in the section on diseases.

1950s

Hurricane Hazel hit the Eastern Shore on October 5, 1954, with 161 kph winds and 45 m waves (*Daily Banner*, October 15, 1954). Farm pastures and agricultural land was destroyed by salt water (*Daily Banner*, October 20, 1954). It was estimated that from 2,200 to 11,000 kg/ha salt was deposited on the fields (*Daily Banner*, November 4, 1954). Captain Amos Creighton, Department of Tidewater Fisheries, surveyed Eastern Shore oyster beds and found little damage (*Daily Banner*, October 22, 1954). The Compass (Maryland Department of Tidewater Fisheries, December, 1954) reported that the grass beds did not seem to be damaged even though the storm was the worst one since August 23, 1933. Within three years after Hurricane Hazel and tropical storms Connie and Diane (1955), aquatic plant populations in the Bay were termed luxuriant (Springer et al. 1958).

Invasion of the upper Chesapeake Bay by Eurasian watermilfoil (*Myriophyllum spicatum*) became apparent in 1954 when plants were discovered in the Gunpowder River (Steenis et al. 1962). By 1958, *Myriophyllum* was becoming a very serious pest plant. Chesapeake Bay Laboratory, Virginia Institute of Marine Science and MBHRL had begun to receive complaints and aid requests from property owners, and sport, civic and commercial groups. Creeks and small rivers where choked with milfoil mats that prevented navigation (Rawls 1964).

Waterchestnut infestation became a serious problem again in 1950 with approximately 72 km of the Potomac River covered with the species. The River and Harbor Act of May 17, 1950, provided authorization for the complete eradication of *Trapa* from the Potomac River and tributaties (Kolessar 1967)

1960s

By 1961, an estimated 20,200 ha of tidewater Maryland were infested with *Myriophyllum*. In the summer of the following year, over 40,500 ha were covered (Rawls 1964). On the Susquehanna Flats alone, autumn surveys in 1960 documented milfoil at 84 percent of the 60 to 80 stations surveyed (Stotts 1961).

In June of 1967, continuous excessive precipitation caused a "wash-in" of muddy water. Though this condition was probably not solely at fault, it seemed to be instrumental in bringing about a noticeable reduction of SAV to scattered pockets of vegetation in the fresh and brackish portions of the Bay (Steenis 1970).

1970s

Hurricane Agnes reached the Chesapeake Bay on June 21, 1972, after an unusually wet winter and spring. Over a three-day period from 19 to 45 cm of rain was dumped on various portions of the Bay watershed. Documented results of the subsequent flooding include impacts to: salinity regimes; sedimentation rates; dissolved phosphorus and nitrogen levels; trace metal and pesticide budgets; dissolved oxygen content; shellfish; aquatic plants; plankton; and benthos. Economic impact was assessed at $43 million distributed mainly on the Western Shore in the shellfishing, tourism and recreation industries in both Maryland and Virginia (Davis 1974).

Bayley et al. (in press) indicated that submerged aquatic plant recovery had been slower than after the storms of the 1950s. Also after Agnes they reported that dominant aquatic plant species in the Susquehanna Flats no longer grew at the deeper depths as during the early 1960s.

DOCUMENTATION SURVEYS

Documentation surveys vary widely as to geographic coverage, season, year, sampling method, data analysis and purpose. The only common denominator is species occurrence.

The two most important variables that affect the individual surveys results are season and data quantification. The time of year in which a survey is performed will greatly affect the results. Some species, for example *Zannichellia palustris*, exhibits two growing seasons, one in the spring and another in the fall. Therefore, a survey performed during July and August, such as the MBHRL Survey, would not document as much *Z. palustris* as a survey done in May or June or in the fall.

In comparing some surveys, data quantification becomes a problem of semantics. The commonly used terms of "abundant", "common" and "rare" sometimes refer to visual estimates and other times refer to volumetric analysis. Visual estimates are highly subjective and at times are based on the percent of a grab sample or the density of a quadrat. Other times the assessment is made according to relative abundance among species.

Survey data is presented in this chapter to the degree of analysis available in the original work. However, conclusions based on the body of information available by combining all surveys has been limited primarily to species occurrence due to the analysis problems just described.

General conclusions and possible trends based on these surveys are presented and analyzed in Chapter 4 in relation to the environmental parameters that are known to affect submerged aquatic vegetation.

Susquehanna Flats Survey, 1958 - present

The first major study of rooted aquatics was conceived in 1957 to study the effects of hurricanes on the Susquehanna Flats. Paul Springer, Robert E. Stewart and Francis Uhler, MBHRL, were requested to investigate what species were present on the Flats. Sampling began during the summer of 1958 along established transect lines covering a total of 37 km (Bayley et al. in press). Sampling was accomplished every 274 to 366 m by dragging grappling plant hooks spaced about 3.05 m apart. Plants were rated as "abundant," "common," "occasional," or "rare" based on visual observations. Table 20 represents the ratings over the 18-year sampling period for the four dominant SAV species found on the Flats.

Though initiated in order to study the effects of hurricanes, this survey was able to document not only population fluctuations of native rooted aquatic species but also the spectacular rise and decline of Myriophyllum spicatum. Results of the survey through 1976 include:

- Eurasian watermilfoil increased in occurrence from 1958 to a peak in 1962.
- Other dominant species remained at fairly constant levels from 1958 to 1961.
- Dramatic declines of wild celery, naiads and elodea were noted in 1962.
- Milfoil decreased after 1962 except for slight increases in 1966 and 1967.
- As milfoil decreased after 1962, other dominant native species increased, returning to near former levels after 8 to 10 years.
- In 1972 all plant populations declined to near zero.
- Plant populations have remained extremely low since 1972.
- Of the average 13.5 species documented from 1958 to 1962, 9 species remained from 1963 to 1965 and 10.5 species were present from 1966 to 1971.
- Potamogeton amplifolius, P. gramineus, and P. nodosus did not return after milfoil declined.

Benthic Survey, 1959 - 1960

In order to determine the importance of SAV and benthos to waterfowl, Stotts (1960) initiated an inventory of Bay benthos in 1959 for correlation with waterfowl numbers. Based on nine established Bay transects (see Figure 35), sampling occurred in autumn and was continued for three years. The transect lines were established in correlation with water quality transects being used by The Johns Hopkins Chesapeake Bay Institute. Samples were taken mainly with a Peterson sampler and station locations were determined by ambient depth. Three samples were taken in both the 0 to 1.5-m and the 1.8 to 3.7-m range.

Dominant plants	Rating category	1958	1959	1960	1961	1962	1963	1964	1965	1966	1967	1968	1969	1970	1971	1972	1973	1974	1975
Myriophyllum spicatum	A^1	0	2	8	27	69	40	30	19	20	20	17	7	14	5	0	0	0	0
	C^1	0	5	20	40	7	10	15	11	21	16	10	15	7	16	0	0	5	3
	O^1	0	18	34	15	6	9	9	9	4	12	9	5	4	6	3	2	8	5
	R^1	2	18	13	18	4	16	12	20	6	11	20	30	19	25	13	23	26	13
	V^2	2	77	173	276	313	224	195	147	157	163	136	113	104	105	18	27	57	32
Vallisneria americana	A^1	59	20	8	9	9	1	1	15	22	26	28	4	8	23	0	0	0	0
	C^1	34	27	24	37	14	15	3	16	21	23	29	21	24	28	0	0	0	0
	O^1	7	34	48	46	34	15	5	4	7	4	7	16	9	10	0	0	0	0
	R^1	0	5	3	12	14	13	8	14	10	0	8	21	13	11	0	0	0	0
	V^2	352	234	203	251	160	92	31	130	175	181	221	132	135	207	0	0	0	0
Najas spp.	A^1	53	46	53	49	5	0	5	5	24	21	14	3	3	9	0	0	0	0
	C^1	13	7	13	17	19	6	11	10	15	18	24	6	9	10	0	0	0	0
	O^1	10	9	12	14	12	3	4	10	9	9	3	6	3	5	0	0	0	0
	R^1	8	4	7	5	12	24	8	20	11	1	14	18	7	9	0	0	1	0
	V^2	279	227	282	280	113	48	69	90	170	152	148	60	52	85	0	0	1	0
Elodea canadensis	A^1	10	0	3	3	1	0	0	3	0	4	20	12	7	7	0	0	0	0
	C^1	15	18	8	16	0	0	3	5	3	17	17	10	14	12	0	0	0	0
	O^1	30	25	24	38	14	1	0	4	3	2	9	15	9	8	0	0	0	0
	R^1	3	14	34	17	12	10	3	5	14	10	11	23	12	16	0	0	0	0
	V^2	148	118	118	153	44	12	12	40	29	81	160	131	100	96	0	0	0	0

[a] Bayley et al. in press.

[1] Number of times species was rated in each category. Differences due to the varying number of stations each year have been normalized to the average number of stations for the duration of the study. "A" = abundant; "C" = common; "O" = occasional; "R" = rare.

[2] "V" is a rating value that gives an estimate of the total amount of plant material according to the following: 4 - abundant, 3 - common, 2 - occasional, and 1 - rare
These values are multiplied by the number of times each category appeared for the species each year.

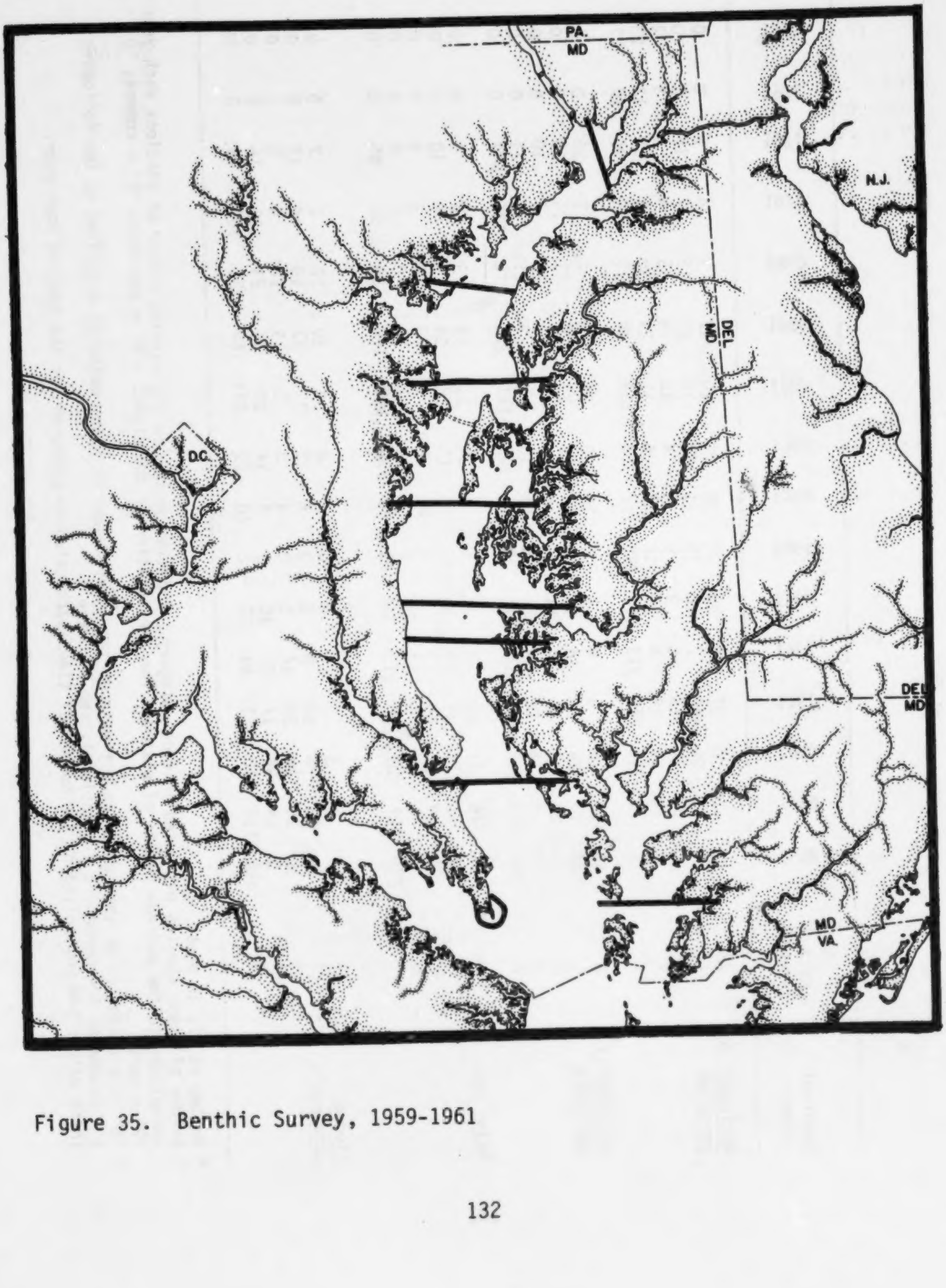

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Figure 35. Benthic Survey, 1959-1961

The survey was discontinued subsequent to the 1961 sampling due to the inadequacy of sampling scale in contrast to the extreme variability of organisms and conditions within the Bay. Table 21 presents percentage occurrence of vegetation based on Stotts' original sample yields.

The limited coverage of the Bay plus the short-term nature of the sampling program do not allow for conclusions as to SAV declines based solely on the Benthic Survey. However, the survey does provide some evidence as to species occurrence especially when correlated with the results of the other surveys. These correlation results are outlined later in this section.

Vegetation Survey, 1967-1969

In order to determine variations in abundance and distribution of SAV in estuarine shoalwaters of Maryland's Eastern Shore, Stotts (1970) initiated a further survey of the Chesapeake Bay in 1967. Established as a pilot study, the survey attempted to determine what methods would be necessary to develop a study that would relate significant changes in SAV occurrence and abundance.

A system of transects was established covering all bays and major rivers. Located approximately one mile apart, these transects ran more or less perpendicular to the shore. From one to three samples were taken on each transect based on available depths, one station fell within each 0.3 to 0.9, 1.2 to 1.8, and 2.1 to 2.4-m depths. Four 3.0 m random drags with a 35-cm cultivator rake head were made at each station and the results for each drag recorded by species and rated as to "abundant," "common," "occasional" or "rare."

A total of approximately 1,000 transects were sampled over the three years. In 1967, about 250 transects were surveyed from the Maryland/Virginia state line north to the head of the Honga River. The following year approximately 375 transects were sampled from Hooper's Island to Tilgham Point. In the final year, about 380 transects were surveyed north from Tilghman Point up into the Elk River. Figure 35 delineates the areas covered each year and Table 22 and 23 shows frequency of occurrence. Survey findings include:

- Massive mortality of SAV occurred north of the Choptank River during the latter part of July and early August.

- A general comparison of SAV abundance indicated that the southerly, more brackish embayments and the fresher bayshores were better producers than the northerly and brackish counterparts, respectively.

- Vegetated samples were located primarily from the 0.3 to 2.0 m range. Vegetation was notably less in the 2.1 to 2.7 m range.

- Causal agents were thought to be heavy rains, high sediment loads and an increase in phytoplankton and zooplankton blooms.

Table 21. Frequency of occurrence of submerged aquatic species, Bentic Survey, 1959-1961[a]

Transect locations and species	Percent of sampling stations		
	1959	1960	1961
Susquehanna Flats	14[b]	47[b]	23[b]
Potamogeton spp.[c]	50	17	22
Vallisneria americana	36	60	69
Najas guadalupensis	36	51	74
Elodea canadensis	0	21	48
Myriophyllum spicatum	0	8	48
Chara sp.	0	19	17
Ceratophyllum demersum	14	0	9
Back River to Fairlee Creek	None	38[b]	25[b]
Myriophyllum spicatum	-	5	16
Vallisneria americana	-	3	16
Najas guadalupensis	-	0	8
Magothy to Chester Rivers	26[b]	47[b]	36[b]
Ruppia maritima	8	11	22
Potamogeton spp.[c]	15	-	-
P. perfoliatus	-	26	28
P. pectinatus	-	6	36
Vallisneria americana	0	4	8
Najas guadalupensis	0	2	33
Zostera maritima	0	2	0
Myriophyllum spicatum	0	0	22
Elodea canadensis	0	0	14
Chara sp	0	0	8
Ceratophyllum demersum	0	0	6
West River to Eastern Bay	None	64[b]	39[b]
Potamogeton perfoliatus	-	9	28
P. pectinatus	-	20	10
Elodea canadensis	-	9	31
Myriophyllum spicatum	-	2	10
Ruppia maritima	-	3	15
Zostera marina	-	5	8
Breezy Point to Choptank River	18[b]	None	None
Ruppia maritima	8	-	-
Potamogeton spp.[c]	8	-	-
Dares Beach to Little Choptank River	None	46[b]	33[b]
Ruppia maritima	-	20	45
Potamogeton perfoliatus	-	9	15
Zostera marina	-	4	6
Elodea canadensis	-	0	3
Patuxent to Honga Rivers	None	72[b]	36[b]
Ruppia maritima	-	17	36
Zostera marina	-	10	50
Point Lookout	9[b]	13[b]	8[b]
Zostera marina	0	75	85
Ruppia maritima	0	38	8
Tangier Sound	16[b]	31[b]	25[b]
Ruppia maritima	38	19	64
Zostera marina	6	3	68

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[a] Vernon Stotts, personal files
[b] Number of stations sampled
[c] In 1959, all species of Potamogeton were grouped together

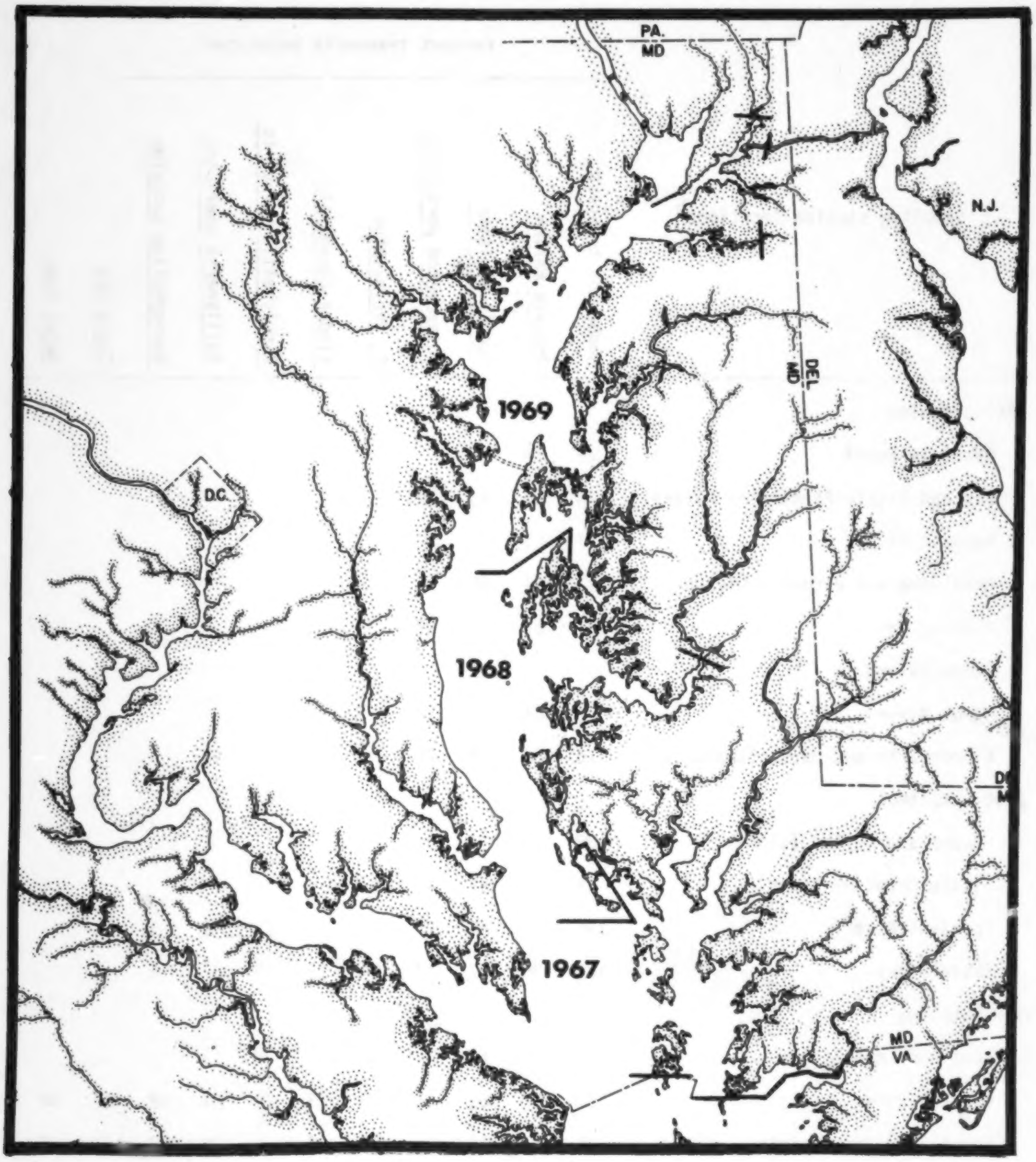
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Figure 36. Vegetation Survey, 1967-1969

Table 22. Frequency of occurrence of vegetated samples, Vegetation Survey, 1967-1969[a]

Sampling station locations[b]	Number of transects	Percent transects vegetated									
		Zostera marina	*Ruppia maritima*	*Potamogeton pectinatus*	*P. perfoliatus*	*Elodea canadensis*	*Zannichellia palustris*	*Vallisneria americana*	*Myriophyllum spicatum*	*Chara* spp.	*Najas* spp.
1967 Sampling											
Pocomoke Sound	19	89	58								
Big and Little Annemessex Rivers	26	96	46	69							
Manokin River	27	85	63								
Nanticoke and Wicomico Rivers	42	21	36								
Fishing Bay	27	56	37	52							
Smith Island	17	78	78								
Honga River	31	84	68	87							
Bloodsworth and Marsh Islands	16	100	81	75							
1968 Sampling											
Hooper and Taylor Islands	39	82	44	59							
Little Choptank River	65	80	97	45	23						
Choptank River	146	32	84	25	91	19	21	1	1		
Eastern Bay	121	36	57	17	69	12	70		2	1	
1969 Sampling											
Kent and Love Points	15	7	47		33						
Chester River	113	1	58	85	26	23	29	21	58	27	30
Howell and Swan Points	44		18	5	20	7		36	23	7	5
Sassafras River	47							34	11	2	2
Elk River	60							5	3	5	

[a] Maryland Wildlife Administration files 1977
[b] Locations correspond to area delineations of MBHRL Survey

General Survey Area	Station Depth	Number of Stations	Zostera marina	Ruppia maritima	Potamogeton perfoliatus	Potamogeton pectinatus	Zannichellia palustris	Elodea canadensis	Valisneria americana	Myriophyllum spicatum
Eastern Bay System	1'-3'	98	0.05	1.10	0.59	0.08	1.75	0.02	-	Tr.
	4'-6'	87	0.53	0.28	1.01	0.17	0.52	0.11	-	0.04
	7'-9'	88	0.35	0.01	0.22	-	0.11	0.12	-	0.03
Kent Island Bayshore	1'-3'	18	-	0.84	0.29	-	-	-	-	-
	4'-6'	16	-	0.84	0.34	-	-	-	-	-
	7'-9'	15	0.01	-	-	-	-	-	-	-
Chester River System	1'-3'	123	-	0.99	1.79	0.11	0.46	0.14	0.34	0.53
	4'-6'	99	0.01	0.14	1.74	0.25	0.07	0.13	0.11	1.02
	7'-9'	99	-	0.01	0.19	-	-	0.01	-	0.04
Kent County Bayshore	1'-3'	43	-	0.37	0.44	0.01	-	0.05	0.90	0.52
	4'-6'	40	-	0.25	0.18	0.01	-	0.08	0.03	0.21
	7'-9'	29	-	-	0.13	-	-	0.08	-	-
Sassafras River System	1'-3'	51	-	-	-	-	-	0.57	0.14	-
	4'-6'	32	-	-	-	-	-	0.10	0.13	
	7'-9'	25	-	-	-	-	-	-	-	-
Elk River System	1'-3'	62	-	-	-	-	-	0.04	0.01	-
	4'-6'	54	-	-	-	-	-	-	-	
	7'-9'	42	-	-	-	-	-	-	-	-

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[a] Assigned grade derived form cumulative abundance ratings of rake drags/depth range divided by number of stations at that depth range. Abundance rating ranged from a high of "4" downward through "3", "2", "1" and "0". Depths have not been corrected to mean low water.

[b] Maryland Department of Game and Inland Fish, inhouse progress report.

Migratory Bird and Habitat Research Laboratory (MBHRL) Survey, 1971 - present

In response to the need for a data base on SAV in the Chesapeake Bay, the U.S. Fish and Wildlife Service Migratory Bird and Habitat Research Laboratory (in Laurel, Maryland) established a summer sampling program of over 600 stations within the Maryland Bay estuary. The objectives of the study were to measure annual trends in abundance, composition and distribution of SAV; and to determine major environmental factors that might impact the vegetation (Kerwin et al. 1975; Munro 1976a, 1976b). Sampling started the summer of 1971 and has continued to the present.

Based on percent acreage of Bay shoal waters (3.2 or less in depth at mean low water), sample station locations were distributed among 26 river system groups (see Figure 37) plus the Potomac River. The delineation of areas based on river systems is not to be interpreted as drainage basins. Rather, boundaries were drawn for ease of sampling and data presentation. Crews returned to the individual established locations each year; however, an estimated error factor of approximately 90 meters parallel to the contour was determined to be acceptable in open areas such as the Susquehanna Flats (Munro, personal communication). At low tide it was occasionally necessary to relocate some stations up to 180 m. Samples were run in triplicate with percent composition by species based on a visual estimate.

In 1972, the MBHRL Survey included not only the Potomac River but also portion of the Virginia section of the Chesapeake Bay. After 1972, these areas were dropped from the survey until 1977 when sampling of the Potomac was resumed. Haramis (1977) provided preliminary results of the Potomac stations which have been included in this document. Available results for Tangier Island and Pocomoke Sound (Virginia) have also been included for 1972. Preliminary results for 1977 for stations located on the Eastern and Western Shores of the Maryland Bay are limited to total vegetated stations by river area. Specific species results will not be available until after June , 1978.

Table 24 shows annual percentage occurrence by river system and Table 25 depicts annual percentage occurrence by plant species. Based on six sampling years, the following results have been noted (Kerwin et al. 1975; Munro 1976a, 1976b):

- Significantly more vegetation was recorded in 1976 than in 1975, though 1976 levels were still only one-half the recorded level for 1971.

- Widgeongrass, horned-pondweed and redhead grass were the most freguently recorded species over the six years.

- Sago pondweed, eelgrass and wild celery continued to be recorded infrequently.

- Concern had been expressed for the Choptank River after the 1975 survey. However, vegetation in the Choptank showed a significant increase in 1976 over the extremely scarce occurrence in 1975.

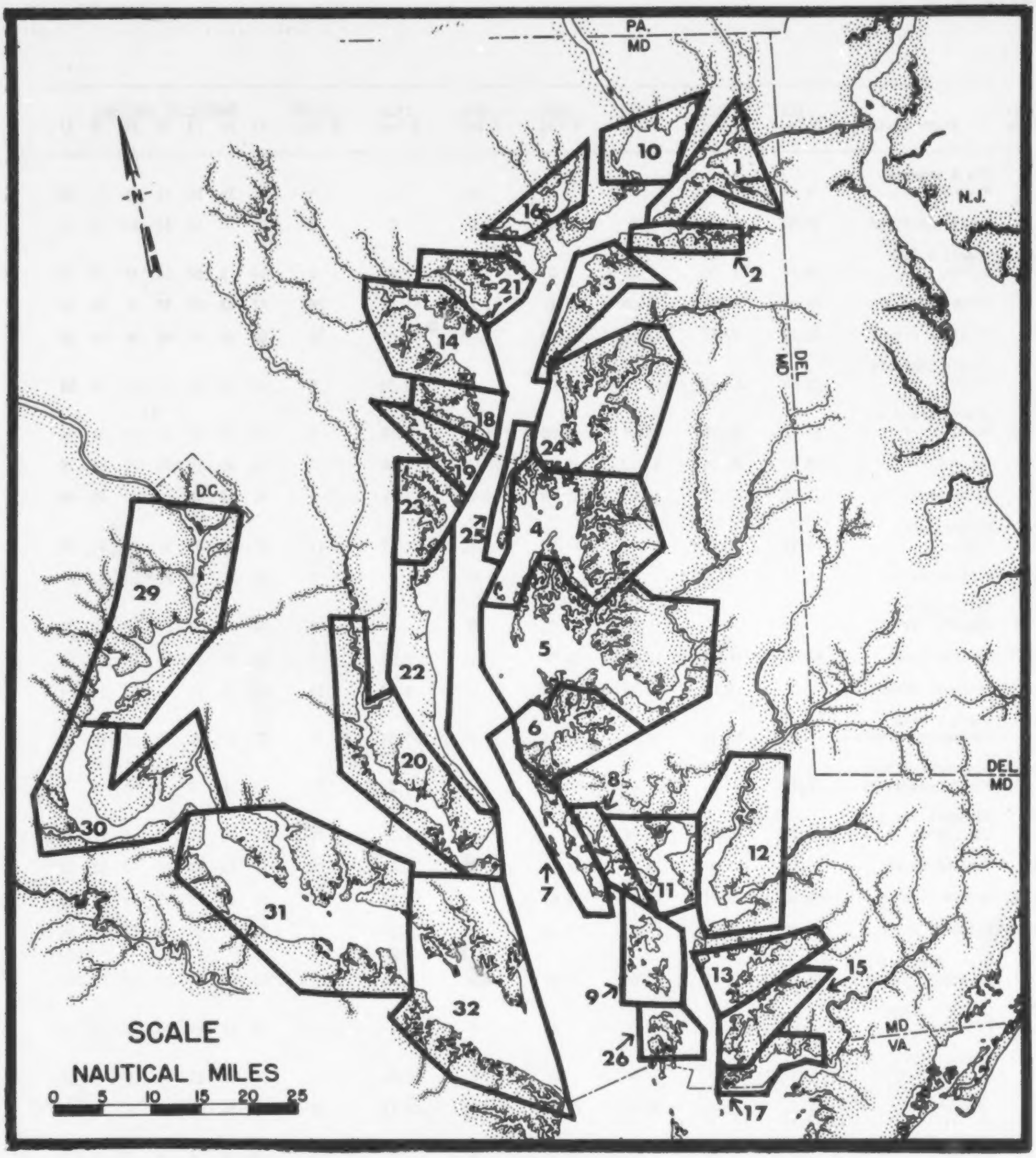
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Figure 37 . River system sampling areas, Migratory Bird and Habitat Research Laboratory Survey, 1971-present

Table 24. Frequency of occurrence of vegetated samples and indicated change by river systems, Migratory Bird and Habitat Research Laboratory Survey, 1971-1976[a]

Area code	River system	1971 % Veg.	1972 % Veg.	1973 % Veg.	1974 % Veg.	1975 % Veg.	1976 % Veg.	1977[b] % Veg.	Number of stations 71	72	73	74	75	76	77
1	Elk & Bohemia Rivers	6.67	0	0	0	0	0	0	15	16	16	16	16	16	16
2	Sassafras River	30.00	0	0	0	0	0	0	10	10	10	10	10	10	10
3	Howell & Swan Points	16.67	0	0	0	0	0	0	12	6	12	12	12	12	12
4	Eastern Bay	34.04	46.51	34.04	36.17	21.74	42.22	28	47	43	47	47	46	45	47
5	Choptank River	35.00	39.66	19.30	27.59	1.72	41.07	25	60	58	57	58	58	56	60
6	Little Choptank River	21.05	21.05	0	0	0	15.79	5	19	19	19	19	19	19	19
7	James Island & Honga River	44.12	35.29	2.94	5.88	5.88	8.82	3	34	34	34	34	34	34	34
8	Honga River	50.00	40.00	13.33	16.66	10.35	17.24	3	30	30	30	30	29	29	30
9	Bloodsworth Is.	37.50	22.73	10.87	11.63	6.98	2.22	4	40	44	46	43	43	45	46
10	Susquehanna Flats	44.44	2.70	0	13.51	11.11	8.57	11	27	37	37	37	36	35	37
11	Fishing Bay	8.00	4.00	0	0	0	0	0	25	25	25	25	24	25	25
12	Nanticoke & Wicomico Rivers	0	0	0	0	0	0	0	30	30	30	31	30	30	31
13	Manokin River	40.00	46.67	13.33	20.00	7.14	6.67	20	15	15	15	15	14	15	15
14	Patapsco River	0	5.00	4.76	9.52		9.52	14	21	20	21	21	0	21	21
15	Big & Little Annemessex Rivers	70.00	60.00	30.00	57.89	33.33	30.00	30	20	20	20	19	18	20	20
16	Gunpowder & Bush River Headwaters	11.11	0	0	0		0	11	9	8	7	9	0	9	9
17	Pocomoke Sound (Maryland)	18.18	10.00	4.76		15.00	9.09	10	22	20	21	0	20	22	22
18	Magothy River	33.33	0	16.67	16.66	-	16.67	25	12	12	12	12	0	12	12
19	Severn River	40.00	20.00	26.67	26.67	-	46.15	20	15	15	15	15	0	13	15
20	Patuxent River	2.00	4.26	0	4.00	0	2.04	2	50	47	50	50	47	49	50
21	Back, Middle & Gunpowder Rivers	13.64	4.55	4.55	4.55	9.09	4.55	9	22	22	22	22	22	22	22
22	Curtis & Cove Points	0	0	0	0	0	0	0	20	19	19	19	6	21	21
23	South, West & Rhode Rivers	0	0	0	0	0	12.50	0	8	10	10	8	8	8	10
24	Chester River	61.11	36.11	26.47	23.52	25.00	25.71	38	36	36	34	34	36	35	36
25	Love & Kent Points	0	0	0	12.50	0	0	0	8	8	8	8	8	8	8
26	Smith Island (Maryland)	64.71	45.46	25.00	35.29	22.22	35.29	24	17	11	12	17	17	17	17
	Total	28.53	20.98	10.49	14.85	8.70	14.97	12	624	615	629	611	552	628	645

[a] U.S. Fish and Wildlife Service Migratory Bird and Habitat Research Laboratory files 1977

[b] Preliminary results (Stotts, personal communication)

Table 25. Frequency of occurrence and indicated change in species of submerged aquatic plants, Migratory Bird and Habitat Research Laboratory Survey, 1971-1976[a]

Species	Percent occurrence					
	1971	1972	1973	1974	1975	1976
Potamogeton perfoliatus	5.29	3.41	4.13	3.43	1.81	2.23
P. pectinatus	1.28	5.69	2.86	1.31	0	0.48
P. pusillus	0.32	0	0.16	0	0	0
Zannichellia palustris	0	2.76	0.16	0.16	0	3.51
Ruppia maritima	14.74	9.92	6.04	9.79	4.69	11.48
Zostera marina	10.26	7.32	1.11	2.45	3.07	1.12
Vallisneria americana	2.72	0.16	0.47	0.82	0.90	0.64
Myriophyllum spicatum	3.85	1.14	0.48	1.14	0.72	0.96
Elodea canadensis	3.04	1.46	0.32	0.49	0.36	0.80
Najas guadalupensis	1.60	0.33	0.95	1.47	0.54	0.32
Ceratophyllum demersum	0.64	0	0	0.16	0	0
Chara spp.	0.32	0.33	0.64	0.49	0	0.16

[a] Migratory Bird and Habitat Research Laboratory files 1977

- Vegetation in the Bloodsworth Island area was at the lowest level recorded during 1976.

- Salinities decreased significantly over the upper Bay averaging:

1971	15.441 ppt
1972	9.663 ppt
1973	10.371 ppt
1974	13.490 ppt
1975	10.956 ppt
1976	8.487 ppt

- Secchi disk readings were taken from 1972 through 1976 and showed a small overall increase, indicating greater light penetration.

- Preliminary results from the 1977 survey indicate that the total number vegetated stations decreased about 2 percent from the 1976 count.

MBHRL data related to individual species are presented in Chapter 1 in the write-ups of the SAV species common to the Bay. The increase in light penetration documented by the MBHRL Survey is not consistent with the commonly held notion that the Bay is becomming more turbid. Chapter 4 discusses the problems and inconsistencies related to assessing turbidity.

Milfoil Survey, 1957-1977

In the late 1950s, in response to a growing concern and a need for information concerning the infestation of Eurasian watermilfoil (*Myriophyllum spicatum*) a network was formulated among various scientific organizations in the Chesapeake Bay area. Over a 20-year period, the various organizations involved in documenting the milfoil story included: Chesapeake Bay Laboratory (University of Maryland), The Johns Hopkins University, U. S. Fish and Wildlife Service Migratory Bird Habitat Research Laboratory, Maryland Wildlife Administration and the Maryland Department of Chesapeake Bay Affairs. Numerous administrative reports, lectures, progress reports and survey data are available through these various organizations. For ease of presentation, these data have been compiled in this report as a general Milfoil Survey.

Though the milfoil work was begun to study only *Myriophyllum spicatum*, other species were documented. *Myriophyllum* infestations had such a negative impact on indigenous SAV that it was necessary to study submerged aquatic species in general. Though the years, many techniques were utilized for sampling. However, for the purposes of this document, only general qualitative results have been included. SAV documentation data from portions of the 20 years of surveying have been compiled by the various organizations into several reports that deal with observations by river system. These observations have been included in the conclusion section of this report and have been correlated with the results of the other surveys.

South River, 1961

Philipp and Brown (1965) studied the distribution of SAV in two transitional areas of the South River for 1961. One area was near the mouth of the river in a small cove on the southeast side of Lime House Cove. The second site was just north of the Route 50 bridge. Samples were taken at both sites along transect lines and both plant species and bottom sediment were noted over an eight year period.

Elodea canadensis dominated the northern site near the head of the river, though relative abundance was considered sparse. Bottom sediment was found to be muck/clay. The south site near the mouth of the river was dominated by *Potamogeton perfoliatus* mixed with some *Ruppia maritima* and *Myriophyllum spicatum*. This latter species and *P. perfoliatus* dominated the deeper areas of the cove (1.5 m maximum). Dense masses of *Elodea nuttalli* and *P. perfoliatus* were observed at the surface. Sediments at this lcoation were mainly sand with some sand/muck.

Philipp and Brown surveyed these two areas in the South River for a single year in order to study species dominance, salinity, temperature and various other water quality parameters. Their results have been included in order to document SAV presence for 1961.

Rhode River, 1966-1973

Southwick and Pine (1975) surveyed SAV in the Rhode River starting with preliminary observations in the early 1960s and continuing with extensive field surveys from 1966 to 1973. Middle and Back Rivers were included for some of the field surveys.

Surveys were made during the summer months, concentrated in June and July, by traversing the shoreline to cover the area within the .6 to 1.2 m depth range at mean low water. Results were plotted on maps for areal interpretation. Table 26 depicts annual abundance for Rhode River in both hectares and indices of abundance (see explanation at bottom of Table 20).

From observations and data over the survey's eight years, *Myriophyllum* and *P. perfoliatus* showed the most pronounced fluctuations in abundance. The latter species plus *Ruppia* and *P. pectinatus* disappeared entirely in 1970 though within the next two years made at least a partial comeback. The eight year period however, showed a general though erratic downward trend for all species.

Bay Bridge Tidal Pond, 1972-1975

From 1972 to 1975, James R. Goldsberry of the Maryland Wildlife Administration surveyed the submerged vegetation of the tidal pool that was created at the east end of the Chesapeake Bay Bridge when the second bridge span was constructed. Covering about 4 ha, the fill area provided an opportunity to observe the natural succession of SAV into an essentially new area. Transects

Year	*Myriophyllum spicatum*		*Potamogeton perfoliatus*		*Ruppia maritima*		*Zannichellia palustris*		*Potamogeton pectinatus*	
	Hectares	Index[b]	Hectares	Index[b]	Hectares	Index[b]	Hectares	Index[b]	Hectares	Index[b]
1966	58.7	180.6	95.1	296.1	7.5	25.6	3.3	10.4	2.9	9.0
1967	0.6	1.4	2.2	6.7	2.0	4.5	2.3	7.2	1.3	4.6
1968	3.8	4.6	34.7	163.5	3.4	11.3	14.5	49.0	2.1	7.4
1969	4.5	13.0	41.9	141.6	2.4	5.7	12.1	37.7	0.4	1.0
1970	0.3	0.3	0	0	0	0	0.8	2.0	0	0
1971	0.7	2.1	0	0	6.1	28.2	2.8	7.0	1.2	6.9
1972	3.9	6.6	3.6	3.6	3.2	6.8	20.7	68.0	9.3	19.8
1973	1.5	2.5	2.1	6.8	1.7	1.8	5.5	15.1	1.8	2.8

[a] Southwick and Pine, 1975

[b] Index of abundance: $ai = \Sigma\, 6(a) + 3(c) + 1(o)$

Where A: = Index of abundance
6 = rating multiplier devoting abundant vegetation
a = hectares of abundant growth
3 = rating multiplier for common vegetation
c = hectares of common growth
1 = rating multiplier denoting occasional vegetation
0 = hectares of occasional growth

were run the length of the pond and samples were taken at depths from 15 to 100 cm. Bottom composition was found to be generally soft muck with occasional sand or rock.

The first year, 1972, the pond yielded *Ruppia maritima*, *Potamogeton perfoliatus* and *Myriophyllum spicatum*. The second year showed no new species invaders. By the third year, *Potamogeton pusillus* and *Elodea canadensis* were found along with the three species represented in 1972. By 1975, no new species beyond the five already present were found.

In the summer of 1977, lush aquatic growth was again present and monitored by the Maryland Wildlife Administration. Significant increases in *Elodea* were noted in July of 1977 along with extensive beds of *P. perfoliatus*, scattered *Myriophyllum* and a trace of *P. pectinatus* (Jayne, personal communication). The increase in the pond SAV that has been observed since 1972 compared to the documented decline in SAV in the Bay deserves further study. The lane that drains into the pond is vegetated with grass and receives maintenance by state crews. Therefore, it is microwatershed where there is little fertilizer input (Bridge Authority personnel, personal communication) but where herbicides (dalopon and amitol-t) have been sprayed for control of *Phragmites* around the pond. Also, it does have high turbidity from carp populations which are capable of stirring up bottom sediments (Stotts, personal communication).

Marshy Creek, 1975

The files of the Maryland Wildlife Administration contain a study of Marshy Creek, south of Kent Narrows surveyed by James R. Goldsberry on July 7, 1975. Documentation was made of the following species: *Potamogeton perfoliatus*, *Myriophyllum spicatum*, *Najas guadalupensis*, *Ruppia maritima*, *Chara* sp., *Zannichellia palustris* and *Elodea canadensis*.

Potamogeton perfoliatus and *M. spicatum* were found primarily in the mainstream of the creek. The inlets off the creek included some of these species but were predominantly populated by *Ruppia*, *Chara* sp. and *Najas*. This study has been included in order to further document species occurrence for the Eastern Bay area.

Eastern Bay Survey, 1976

During the summer of 1976, George H. Fenwick, The Johns Hopkins University, surveyed SAV in the Eastern Bay. The survey was part of a joint program initiated by the Chesapeake Bay Foundation, The Johns Hopkins University waterfowl research group and the Youth Conservation Corps.

The survey was based on a system of random transects located about 122 shoreline meters apart and extending about 55 m perpendicular to the shore. Sampling stations were located at seven intervals along the 55 m transects. A total of 2,192 stations were sampled in the Wye River and tributaries; Miles River and tributaries; Kent Island; Rich Neck, including Claiborne Harbor; and from Kent Narrows Bridge to Bennett Point. The survey spanned the time between June 28 and August 27, 1976. The various parameters analyzed included: percent occurrence, biomass, plant height, substrate and visual estimates of ground cover.

All submerged vascular species of concern to this technical document were found in the study area, with the exception of Chara spp. Results of the survey include:

- 57.4 percent of the total number of stations were vegetated
- 48.3 percent of the total stations had 10 percent or more ground cover
- 24.2 percent of the total stations had abundant cover
- The Wye River was the most heavily vegetated area studied
- Ruppia maritima generally was considered to be dominant, occupying 52.7 percent of the total biomass
- P. pectinatus was determined to be least common, with 4.0 percent of the total biomass
- Zannichellia died back in mid-July
- P. pectinatus was observed to die back in mid-August
- Ruppia increased in occurrence over the survey time span

This survey has been continued by the same joint effort though results are not available at this time for 1977.

Chesapeake Bay Center for Environmental Studies (CBCES) - 1976-1977

CBCES (Smithsonian Institution) is presently studying the decline of submerged aquatic vegetation in selected areas within the Chesapeake Bay (Correll et al. 1976a, 1976b, 1977). The study is based on testing correlations between plant die-offs, turbidity and herbicides. Station locations in the Rhode, Choptank and Severn Rivers and the Poplar Island area are being sampled for SAV populations, phytoplankton levels, turbidity of surface waters, suspended particulates in surface waters, bottom sediment characteristics and herbicide levels. Study objectives include analyzing the filtered water, suspended particulates and bottom sediments for atrazine, alachlor, linuron and trifluralin. Specific results of the field and bioassay work with Zannichellia palustris and herbicides are discussed in Chapter 4.

Field surveys have documented SAV populations at established station locations throughout the four areas for 1976 and 1977. Results from the 1976 survey have been included in the conclusion section of this chapter.

Virginia Institute of Marine Science Surveys

Submerged aquatic vegetation in the waters of the Virginia portion of the Chesapeake Bay is essentially the history of Zostera marina. Several other species have been documented; however, the majority of the available information on Virginia waters has resulted from the work of Robert Orth (VIMS) who has limited his research primarily to Zostera. The following section was written by Orth and is submitted here as documentation of the history of Z. marina in Virginia.

Robert Orth. In Virginia, eelgrass has been abundant on the Western Shore in the York, Rappahannock and Piankatank Rivers, Mobjack Bay and its tributaries, Back and Poquoson Rivers, Poquoson Flats area, Fleets Bay to the Potomac River and on the Eastern Shore from Cape Charles to Tangier Sound. The first indications of a major decline came in the fall of 1973 (eelgrass had been abundant that summer) though some local people reported eelgrass declined in the low salinity regions of the estuary in 1972 after Tropical Storm Agnes. By 1974, virtually all the eelgrass in the York, Rappahannock and Pinakatank rivers was gone with significant declines in other areas (Table 27, Figure 38 and 39). (Eastern Shore was not surveyed in 1974) (Orth and Gordon 1975; Orth 1976).

Table 27. Estimated total coverage of submerged aquatics for major Virginia sections of the Chesapeake Bay, 1971 and 1974[a]

Major Areas	1971 (ha)	1974 (ha)
Back River	144.9	211.9
Plum Tree Island area	868.2	466.5
Poquoson Flats	233.8	59.9
Poquoson River	66.1	111.7
Crab Neck and Goodwin Islands	306.7	219.4
York River	493.1	140.8
Mobjack Bay	1293.9	1592.6
New Point Comfort to Pototo Neck	168.4	233.0
Gwynn's Island area	266.8	43.3
Piankatank River	306.9	21.2
Rappahannock River	699.6	3.6
Fleets Bay	---	197.6
Hampton area	---	7.2

[a] Orth and Gordon, 1975

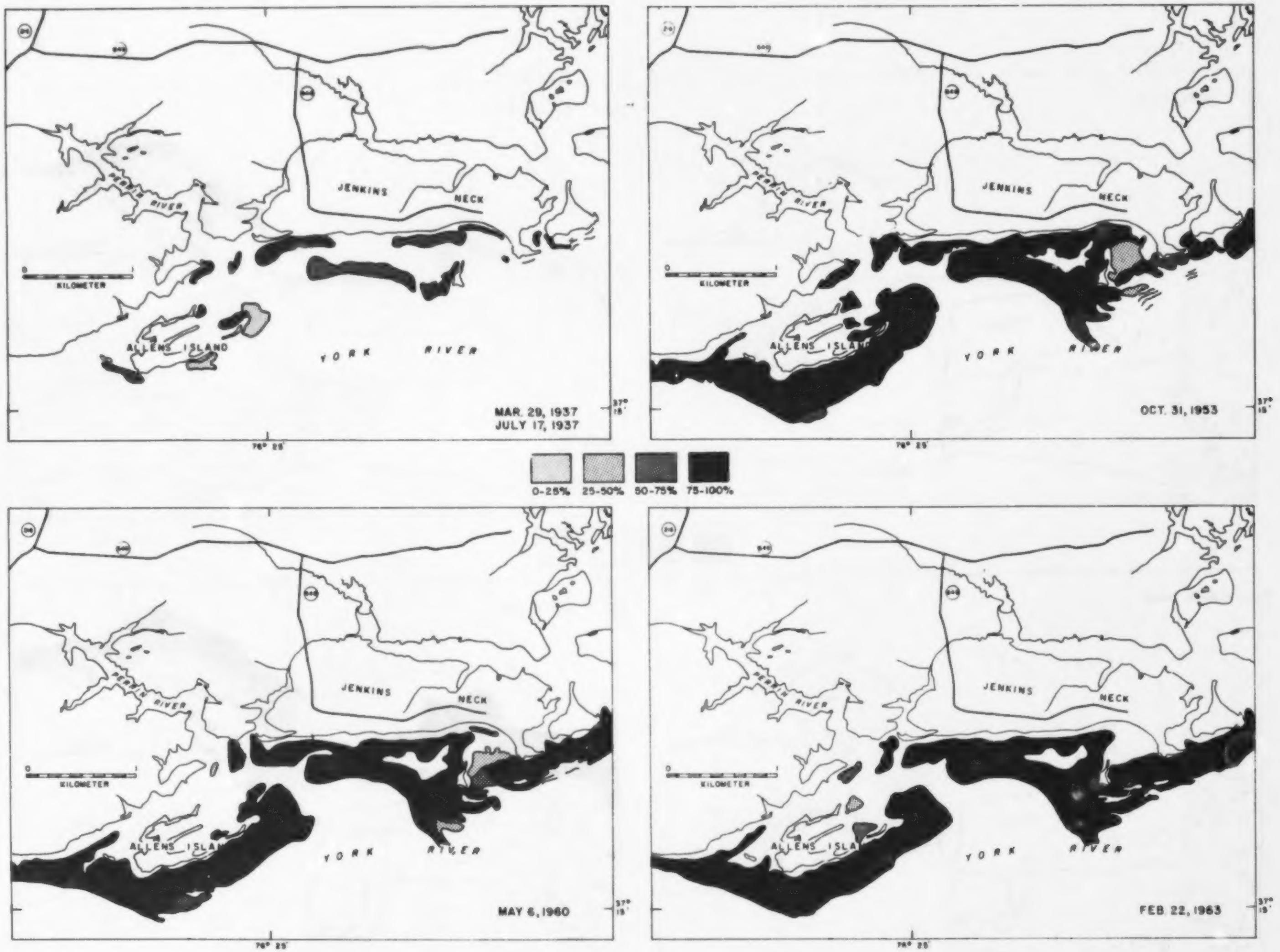
JENKINS
NECK
PERRIN RIVER
ALLENS ISLAND
YORK RIVER
KILOMETER
0
1
MAR. 29, 1937
JULY 17, 1937
OCT. 31, 1953
MAY 6, 1960
FEB 22, 1963
76° 25'
37° 15'
0-25%
25-50%
50-75%
75-100%

Figure 38. Distribution of Zostera marina, Lower York River, 1937-1977

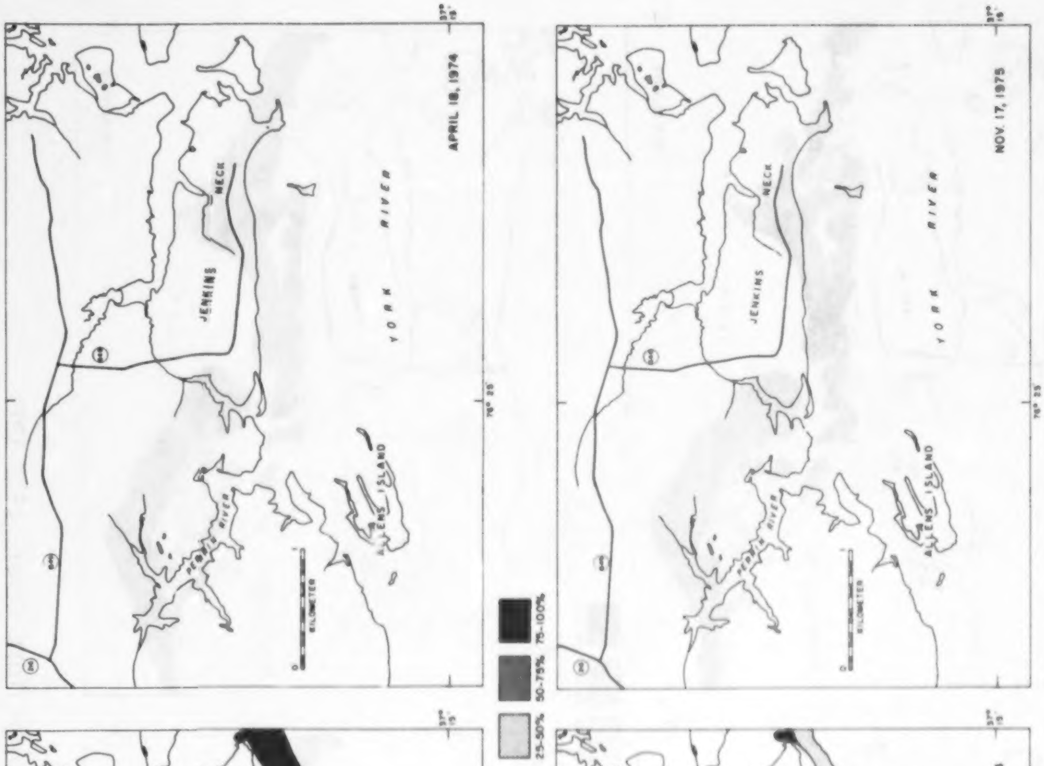
OCT. 12, 1971
APRIL 18, 1974
NOV. 23, 1974
NOV. 17, 1975
JENKINS
NECK
YORK
RIVER
PERRIN RIVER
ALLENS ISLAND
KILOMETER
0
76° 25'
37° 15'
0-25%
25-50%
50-75%
75-100%

Figure 38 (continued)

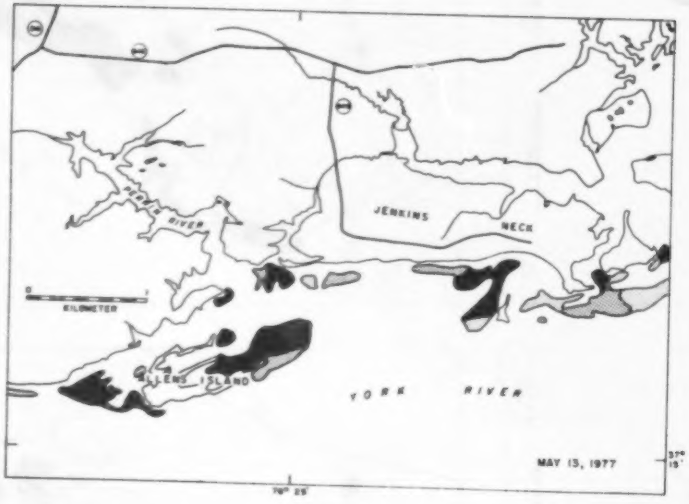
PERRIN RIVER
JENKINS
NECK
0
1
KILOMETER
ALLENS ISLAND
YORK RIVER
MAY 13, 1977
76° 25'
37° 15'

Figure 38. (continued)

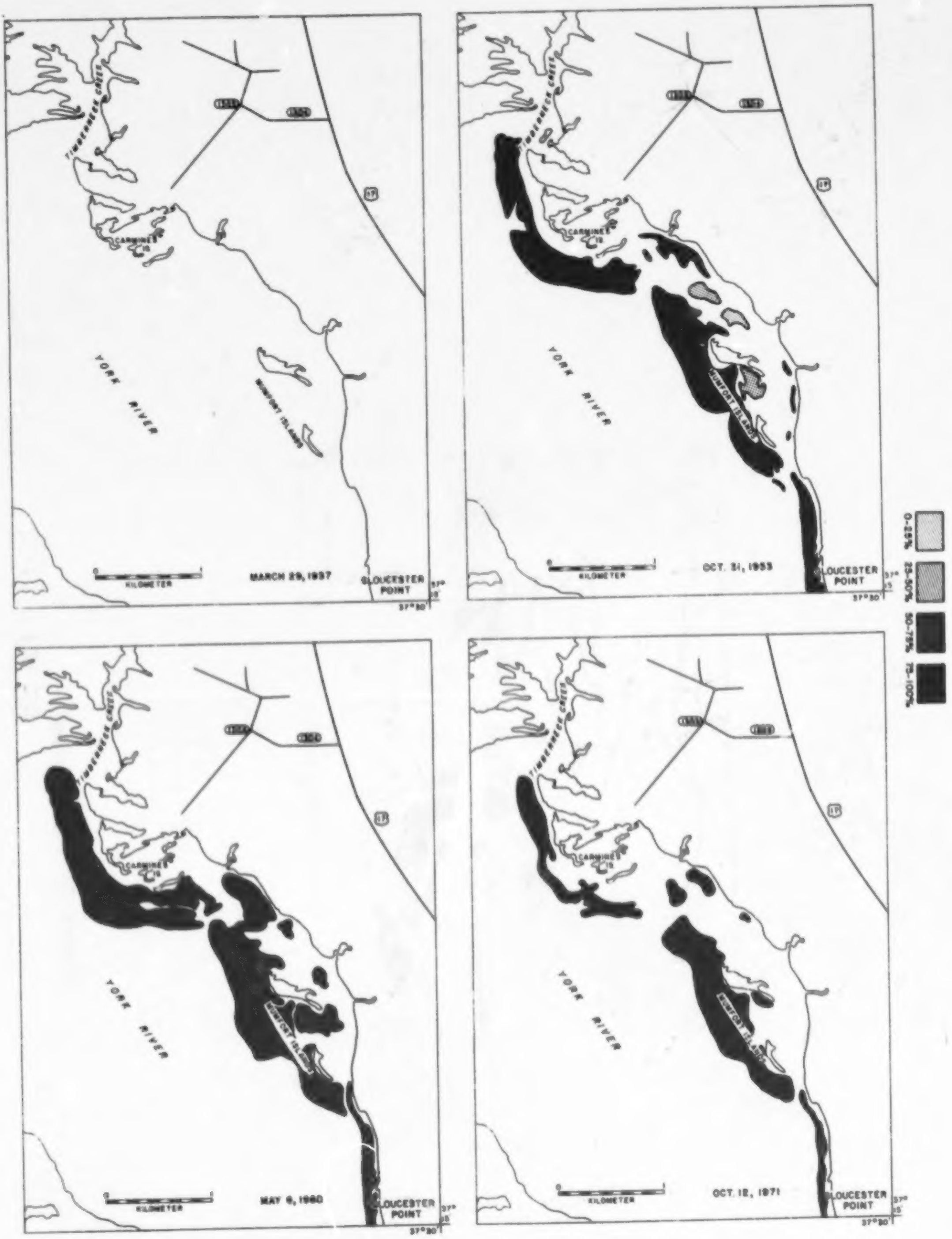
TIMBERNECK CREEK
CARMINES IS
YORK RIVER
MUMFORT ISLANDS
17
KILOMETER
MARCH 29, 1937
GLOUCESTER POINT
37°15'
37°30'
TIMBERNECK CREEK
CARMINES IS
YORK RIVER
MUMFORT ISLANDS
17
KILOMETER
OCT. 31, 1953
GLOUCESTER POINT
37°15'
37°30'
0-25%
25-50%
50-75%
75-100%
TIMBERNECK CREEK
CARMINES IS
YORK RIVER
MUMFORT ISLANDS
17
KILOMETER
MAY 6, 1960
GLOUCESTER POINT
37°15'
37°30'
TIMBERNECK CREEK
CARMINES IS
YORK RIVER
MUMFORT ISLANDS
17
KILOMETER
OCT. 12, 1971
GLOUCESTER POINT
37°15'
37°30'

Figure 39. Distribution of Zostera marina, Middle York River, 1937-1975

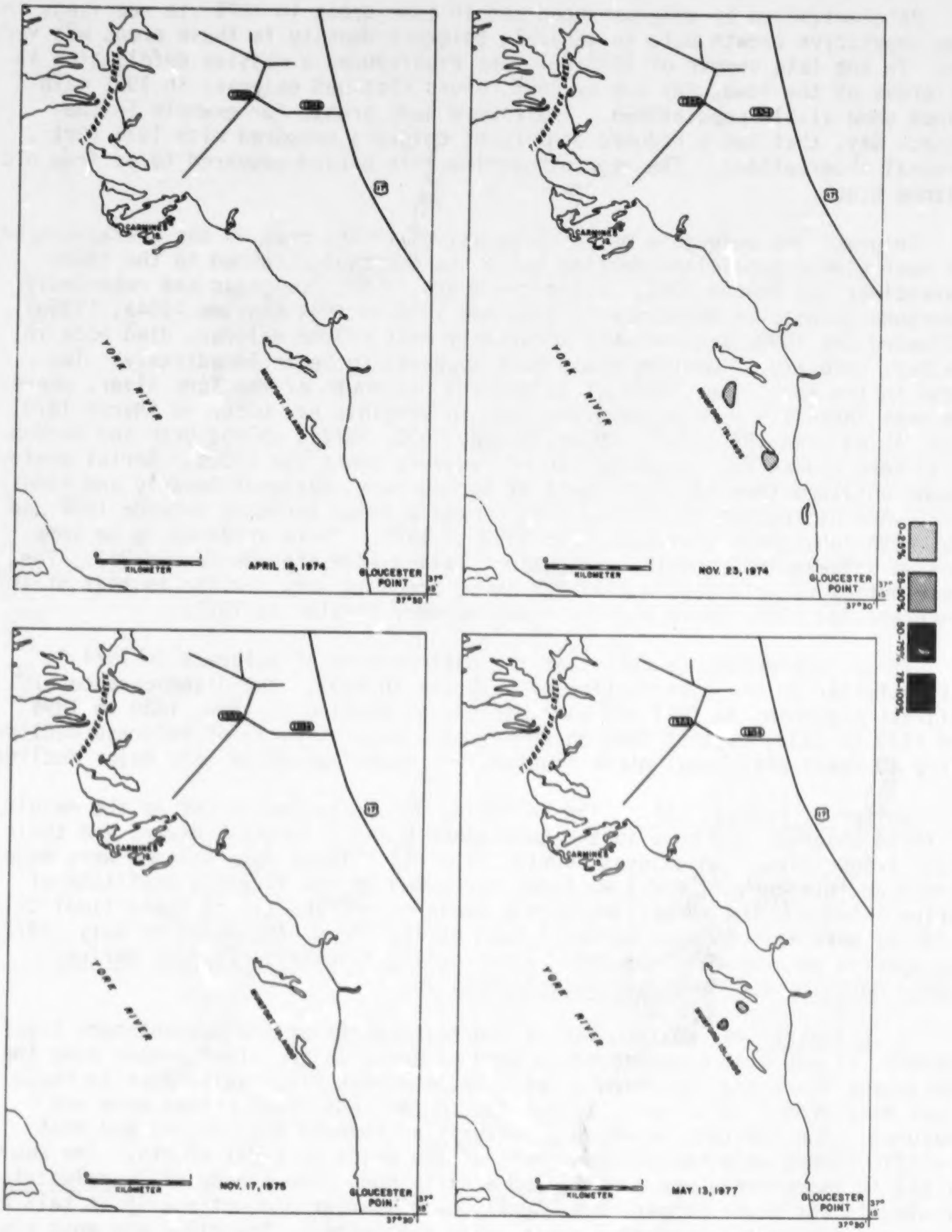
TIMBERNECK CREEK
CARMINES IS.
YORK RIVER
MUMFORD ISLANDS
17
KILOMETER
APRIL 16, 1974
NOV. 23, 1974
NOV. 17, 1975
MAY 13, 1977
GLOUCESTER POINT
37° 15'
37°30'
0-25%
25-50%
50-75%
75-100%

Figure 39. (continued)

Recolonization by eelgrass occurred in some areas in 1974 via seedlings and some vegetative growth but, in general, eelgrass density in these areas was very low. In the late summer of 1975 eelgrass experienced a massive defoliation in all areas of the lower Bay but by 1976, areas that had eelgrass in 1975 maintained some viable populations. There were some areas, for example in the Mobjack Bay, that had a reduced density of eelgrass compared with 1975 (Orth, personal observation). The regrowth during this period appeared to be from old rhizome stock.

Eelgrass has undergone major fluctuations in the past in the Chesapeake Bay, the most widely publicized decline being the one that occurred in the 1930s (Burkholder and Doheny 1968; Cottam and Munro 1954). Eelgrass had reportedly undergone changes in abundance in 1854 and 1889 to 1894 (Cottam 1934a, 1935b). Following the 1930s decline when apparently most of the eelgrass died back in the Bay, recovery of many eelgrass beds appeared to begin immediately. Two areas in the York River, Mumfort Island and the mouth of the York River, where the most intensive work on eelgrass beds in Virginia has occurred (Marsh 1970, 1973, 1976; Orth 1971, 1973, 1975a, 1975b, 1976, 1977a, 1976b; Orth and Gordon 1975) were chosen for documentation of recovery since the 1930s. Aerial photographs obtained from the Department of Agriculture, National Oceanic and Atmospheric Administration (NOAA) and VIMS showed a major increase between 1937 and 1953 with subsequent increases from 1953 to 1971. There appeared to be some decline between 1960 and 1971 at Mumfort Island (see Figures 38 and 39). The distribution of eelgrass in 1977 for these areas is very similar to that of 1974. (Coverage for 1976 showed the distribution very similar to 1977).

It is interesting to note that the distribution of eelgrass in 1974 is quite similar to the distribution of eelgrass in 1937. The disappearance of eelgrass beginning in 1973 and past periods of decline in 1854, 1889 to 1894 and 1931 to 1932, suggest that there may be a natural cycle of eelgrass decline every 40 years with local minor fluctuations superimposed on this major decline.

Walter I. Priest, III. The following documentation of SAV is the result of field observations made in the Rappahannock and Piankatank Rivers and their tidal tributaries in Middlesex County, Virginia. These observations were made during an inventory of tidal wetlands performed by the Virginia Institute of Marine Science. The shorelines of the above rivers and all of their tidal tributaries were investigated by small boat during the months March to July, 1977. The species encountered included: *Zannichellia palustris*, *Zostera marina*, *Ruppia maritima* and *Potamogeton* (*epihydrus* ?).

Z. palustris was most common in the tributaries of the Rappahannock River. However, it was also encountered in several areas in the river proper plus the Piankatank River and its tributaries. Rappahannock River salinities in these areas vary from 7 to 13 ppt. Salinities in the individual creeks were not measured. The habitats in which *Z. palustris* produced the largest and most prolific stands were the shallow areas at the heads of tidal creeks. The substrate in these areas was usually loose silty mud. The stands of *Z. palustris* growing in the river proper were usually much sparser and not nearly as tall as those growing in the headwater areas. The substrate in the river was most always sand. The localities of occurrence are as follows and unless otherwise noted the observed distribution was restricted to the heads of the creeks indicated (see Figure 40):

Mason's Mill Swamp
Mud Creek
Parrotts Creek
Weeks Creek
Lagrange Creek
Urbanna Creek
Whiting Creek
Meachims Creek, west of mouth in river
Greys Point, in river
Locklies Creek, near mouth
Parrott Island, in river
Bush Park Creek
Piankatank River between Horse Pt. and Glebe Neck
My Lady's Swamp near head on Piankatank River

The distribution of *Ruppia maritima* was apparently restricted to the lower reaches of the Rappahannock and Piankatank Rivers. Several large stands were observed near the mouth of Sturgeon Creek. It was also found growing in monospecific stands and mixed with *Zostera marina* on the offshore flats from the mouth of Broad Creek around Stingray Point and up to the mouth of Jackson Creek near the mouth of the Piankatank River. In addition, *R. maritima* was found growing in several tidal ponds along the Rappahannock River including one near the mouth of Mill Creek and the two east of Duck Pond.

Zostera marina was observed in dense stands and in association with *R. martima* on the offshore sandflats from Broad Creek around Stingray Point to Jackson Creek. A broad floating-leaved pondweed tentatively indentified as *Potamogeton* (*epihydrus* ?) was encountered at the heads of Mason's Mill Swamp and LaGrange Creek near the upper limits of tidal influence in essentially fresh water.

Herbarium Survey

Herbaria in the vicinity of the Chesapeake Bay have provided some information as to historic occurrence of SAV. The data does not allow for interpretation as to abundance or absence, but does indicate SAV presence as of the specimen date. Documentation pertaining to the dominant submerged aquatic species in the Bay was collected from the following herbaria:

University of Maryland, College Park, Maryland
Towson State University, Towson, Maryland
Smithsonian Institution, Washington, D.C.
Clyde Reed, (private collection)
Chesapeake Biological Laboratory, Solomons Island, Maryland
U. S. Fish and Wildlife Service Migratory Bird Habitat Research Laboratory, Laurel, Maryland

The Smithsonian Institution herbarium contains the major holdings of submerged aquatic species in the Chesapeake Bay area. Virginia herbaria were not included in the survey due to time and distance problems.

Documentation of plant specimens has been organized along the format of the river system groupings of the MBHRL sampling program. These observations are included in the conclusion section at the end of this chapter.

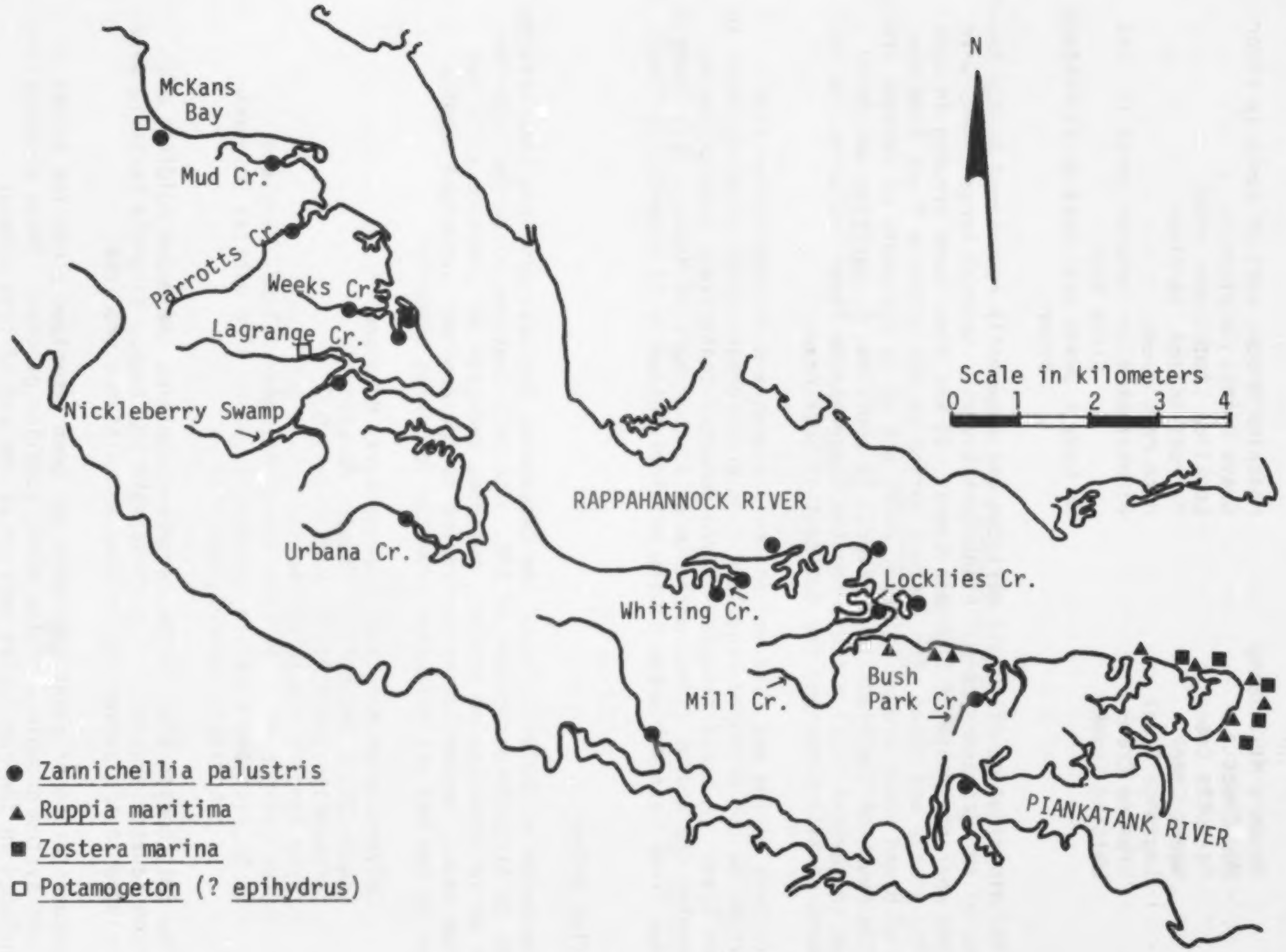
McKans Bay
Mud Cr.
Parrotts Cr.
Weeks Cr.
Lagrange Cr.
Nickleberry Swamp
Urbana Cr.
RAPPAHANNOCK RIVER
Whiting Cr.
Locklies Cr.
Mill Cr.
Bush Park Cr.
PIANKATANK RIVER
N
Scale in kilometers
0
1
2
3
4
● Zannichellia palustris
▲ Ruppia maritima
■ Zostera marina
□ Potamogeton (? epihydrus)

Figure 40. Distribution of submerged aquatic plants in tidal areas of Middlesex County, Virginia, 1977

Personal Communications

Just about everyone who has lived around the Chesapeake Bay in his/her life has a remembrance of submerged aquatic vegetation. Often, the story is of an unpleasant association: fouling of boat props, overgrowth of swimming areas, noxious odors or decaying wind rows, etc. But many of the associations have been pleasant: increased number of waterfowl for hunting, good fishing and crabbing areas, etc. Thus, the relationship between humans and SAV has been erratic and personal, a pleasure/pain existence.

Time does not permit the research necessary to extensively document this relationship with first hand accounts. Also, such documentation would be of questionable value to a scientific data base on SAV in the Chesapeake Bay since memories are often faulty. However, in the course of the research effort for this technical document several knowledgeable and aware individuals have been contacted and interviewed. Summaries of these interviews are included in this section.

Francis, Uhler, MBHRL, January 10, 1977. Mr. Uhler speaks at length of his 50-plus years of observations in the Chesapeake Bay area. His ready recall of observations made in the late 1920s is as instantaneous as his 1977 recall. Though most of his work has been along the Potomac River, his research has taken him over most of the Maryland portion of the Bay at one time or another. Uhler has amassed thousands of photographic slides taken since the 1930s. These slides are presently being indexed and hopefully will be available sometime in the future. The historic documentation that could be provided by Uhler's slide collection would be invaluable to a general data base on SAV.

Some observations made by Uhler concerning his many years in the area include:

- In Goose Bay (Port Tobacco River), Potomac River beds of *Vallisneria americana* were visible in 1976-1977 but greatly reduced from beds present in the area in 1927. Marina dredging and algal blooms have greatly changed the area. Port Tobacco River marinas have drastically impacted the *Vallisneria* beds.

- In 1976, windrows of *Vallisneria* were seen along the banks of Popes Creek and Persimmon Point, Potomac River.

- Luxuriant growths of *Ruppia*, *P. pectinatus* and *Vallisneria* were seen in the Potomac River prior to the 1931 drought. Vegetation returned only in pockets.

- In the Blackwater Wildlife Refuge and Honga River area *P. pectinatus* and *Zostera marina* were seen prior to Agnes. Species of *Chara* common to the area included *C. geobularis*, *C. braunii*, *C. zeylanica* and *C. hornamanii.*

- Heavy beds of *P. pectinatus* and *Zostera* were decimated by a bryozoan 4 to 5 years ago in Trippe Bay.

- In the Wye Narrows where P. pectinatus was abundant 5 years ago, there is none.

- Canvasback stomach studies from the Nanticoke River presently show only invertebrate material. The area is presently too turbid for SAV.

- Gullet and stomach studies on Canvasbacks have shown that this species has switched to an animal diet of Rangia, Macoma and Corbicula

Earl Abey, Waterman, May 27, 1977. Earl Abey has lived at Green Point for over 50 years; his front porch is approximately 9 meters off the Choptank River. "Up until 1970, the river in front of the house was like a wheat field, it was so green," remembers Abey. Most of the Choptank was like that, with seasonal SAV blooms protruding from the surface. In September, the vegetation normally broke off at the rhizomes and floated in large masses to shore.

Dr. Eugene Cronin went out on the Choptank with Abey in 1944. At this time Abey remembers extensive vegetation stands. Up until 1970, he used to cut channels for his boats to get out, but today he goes out to the docks and sees only a sprig or two.

Compared to other areas of the Choptank, Green Point is secluded and therefore protected. During the summer an algal and scum layer would stagnate above the rooted aquatics. Abey feels that with the decline in vegetation, he receives more wave activity near his docks.

Abey attributes the vegetation decline to herbicides runoff from farmland. He remembers an example of herbicides affecting vegetation control some 20 years ago when someone put a gallon of herbicide in a pond near his house. Vegetation has never returned.

Even though rooted aquatics have declined, Abey finds that crabs are still abundant in the deeper waters. However, he does not know how the vegetation loss will affect the seafood population in the long run.

CONCLUSIONS

In order to synthesize the diverse data sources (surveys, specimens, personal communications, etc.) into a more logical and simplistic structure the river system delineations utilized in the MBHRL study will serve as an organizational format. This areal partitioning of the Maryland portion of the Chesapeake Bay has been previously shown in Figure 37. Data for the Potomac River includes both the Maryland and Virginia sides of the River. Virginia vegetation documentation is listed by major river systems. Table 28 provides a source key for following historic documentation tables.

Area 1. Elk and Bohemia Rivers

The first documentation of SAV in the Elk and Bohemia Rivers dates back to 1955 herbarium specimens of P. perfoliatus, Vallisneria and Elodea

Table 28. Historic documentation key

(A)	Anderson et al. (1967)
(B)	Benthic Survey
(CB)	Chesapeake Bay Center for Environmental Studies
(E)	Elser (1969)
(EC)	Ecological Analysts, Inc. (Pine, personal communication)
(F)	Eastern Bay Survey (Fenwick, unpublished)
(G)	Marshy Creek Survey
(H)	Herbarium specimen
(HP)	Horn Point Environmental Laboratories
(M)	Milfoil Survey
(MB)	MBHRL Survey (Kerwin et al. 1975; Munro 1976a, 1976b)
(O)	Orth, VIMS
(P)	Pine, personal observation
(P&P)	South River Survey (Philipp and Brown 1965)
(RS)	Stewart (1962)
(S)	Susquehanna Flats Survey (Bayley et al. in press)
(SO)	Southwick (1967-1969)
(S&P)	Rhode River Survey (Southwick and Pine 1975)
(U)	Uhler, personal observation
(V)	Vegetation Survey
(VS)	Stotts, personal communication
(WP)	Priest, VIMS

(see Table 29). Documentation through the 1960s and until 1971 indicates the continued presence of five species. However, since 1971, no plants have been observed by the MBHRL Survey. P. perfoliatus has not been documented since 1962 and Elodea since 1969. Milfoil Survey personnel (1970-1971) found SAV growth to be diversified but sparse and limited to indentations in the lower part of the Elk River and the upper reaches of Veazey Cove, Bohemia River. Channel deepening in the Elk River and C & D Canal resulted in a heavy silt load that was readily apparent in August of 1969. No vegetation was observed in the turbid waters of Cabin John Creek below the Bohemia River.

Table 29. Historic documentation of SAV by decades, Elk and Bohemia Rivers, Maryland (Area 1)

Species	1950	1960	1970
Potamogeton perfoliatus	1955(H)	1962(H)	
Ceratophyllum demersum		1961,1969(M)	1970-1971(M)
Myriophyllum spicatum		1961,1967-1969(M) 1963,1968(H) 1969(V)	1970-1971(M)
Zannichellia palustris			1970-1971(M)
Vallisneria americana	1955(H)	1961,1969(M) 1969(V)	1970-1971(M) 1971(MB)
Elodea canadensis	1955(H)	1961,1969(M)	
Najas spp.		1961,1969(M) 1968(H)	1970-1971(M)
Chara spp.		1969(V)	

Area 2. Sassafras River

Herbarium specimens date back to 1947 and 1952 for the Sassafras River (see Table 30). During the 1960s, survey teams documented a total of seven species. Elser (1969) reported extensive stands of Myriophyllum, Ceratophyllum, Vallisneria, P. crispus, Elodea and Najas in the Sassafras River and its tributaries in the early 1960s. By 1966, vegetation seemed to be limited to the tributary mouths and the lower central portions. Beds have previously been seen in intertidal areas plus the upper parts of the tributaries and from the shore out 6 to 15 m. In 1967, Elser observed what he termed the "Rhode River Evanescence" when only a few sparse SAV beds were found. Vallisneria and Ceratophyllum displayed a growth resurgence after the decline in Myriophyllum; however, abundance levels never seemed to reach prior levels.

Table 30. Historic documentation of SAV by decades, Sassafras River, Maryland (Area 2)

Species	1940	1950	1960	1970
Potamogeton perfoliatus	1947(H)	1952(H)	1968-1969(M)	
Myriophyllum spicatum			1960s(E) 1961(M) 1965(H) 1969(V)	
Vallisneria americana		1952(H)	1960s(E) 1961,1969(M) 1969(V)	1970-1971(M) 1971-1974(MB)
Elodea canadensis		1952(H)	1961,1969(M)	
Najas spp.		1952(H)	1960s(E) 1961, 1969(M) 1969(V)	1970-1971(M) 1971(MB)
Chara spp.			1969(V)	
Ceratophyllum demersum			1960s(E) 1961,1968-1969(M)	

Experimental herbicidal control measures were used in Revuques Pond and Shellcross embayments in 1968 for *Myriophyllum* control. In Revuques Pond, the experiment was considered to be 95 percent effective and resulted in dominant growth of *Najas* in 1969. In the Shellcross embayment, *Myriophyllum* control resulted in the dominance of *Vallisneria* in 1969.

The 1970-1971 Milfoil Survey found *Myriophyllum* to be dominant in several embayments along the southern shore of the Sassafras River. Some resurge in growth of *Vallisneria* was noted in Shellcross embayment and along the northern shore. Nothing but sparse vegetation had been observed in Cox, Freeman Island, Woodland and Dyer Creeks since 1966. In 1967, the water in these creeks was observed to contain a heavy silt load and plankton blooms. In Turner, Money and Foreman Creeks, the water was also clouded by a heavy silt load and only scattered SAV beds were noted.

Area 3. Howell and Swan Points

A *Ruppia* herbarium specimen from 1907 is the first documented SAV species listed for Howell and Swan Points (see Table 31). By the 1960s, three survey terms had documented nine aquatic species. Milfoil Survey personnel surveyed Stillpond Creek and found sparse growths of *Vallisneria* in 1970, substantial *Vallisneria* in 1971 and a seeming reinvasion of *Myriophyllum* during 1970 and 1971. The MBHRL Survey found good growths of *Vallisneria* and traces of *Myriophyllum* in Stillpond Creek in 1971. There has been no documentation of any vegetation in Stillpond Creek or in the rest of Area 3 since 1971 with the exception of Stotts who noted dense beds of what may have been milfoil in the lower portion of the creek. These beds were noted on November 14, 1977, during an aerial survey of Canada geese.

Table 31. Historic documentation of SAV by decades, Howell and Swan Points, Maryland (Area 3)

Species	1900	1940	1960	1970
Potamogeton perfoliatus			1961(M) 1965-1969(M) 1969(V)	
Ruppia maritima	1907(H)		1965,1969(M) 1969(V)	
Myriophyllum spicatum			1961(B)(M) 1965,1969(M) 1969(V)	1971(MB) 1970-1971(M)
P. pectinatus			1969(V)	
Vallisneria americana		1947(H)	1961(B)(M) 1965,1969(M) 1969(V)	1971(MB) 1970-1971(M)
Elodea canadensis			1965(M) 1969(V)	
Najas spp.		1947(H)	1961(B) 1969(V)	
Chara spp.			1969(V)	
Ceratophyllum demersum			1969(M)	

Area 4. Eastern Bay

Percentages figures from the MBHRL Survey showing number of stations with vegetation indicate a fairly steady total population over the six years (see Table 24). Ruppia remained dominant and a strong resurge in growth seemed to have taken place between the 1975 and 1976 summer samplings. Trace amounts of P. perfoliatus, Myriophyllum and Zannichellia were found in 1976 and P. pectinatus and Elodea were documented in occasional patches.

Frank Pine, then a graduate student at The Johns Hopkins University, surveyed the Miles River from 1971 through 1974 (Pine, personal communication). Ruppia, P. pectinatus, P. perfoliatus and Zannichellia were documented for all four years around Claiborne, Rich Neck and Long Point. All species occurred throughout the four years though a definite decline in abundance was noted by 1974.

Pine surveyed Parsons Island in Eastern Bay in May, 1974, and found P. pectinatus to be extremely abundant. The beds seemed to be at absolute maximum production with some plants up to 1.8 m in length. By 1975, vegetation appeared to be considerably less. In August and September, 1974, Pine surveyed the Horse Head Farm area. Populations were mixed: P. perfoliatus was the most common throughout the shorelines; Ruppia was common to occasional; and Elodea was found occasionally. Myriophyllum was determined to be locally abundant to occasional on the north side of the island. Zannichellia was not seen, probably due to its normal die-back in July. *P. perfoliatus* was observed to have completed its seed production and started to decline. Ten species of SAV have been documented continuously from 1961 to 1976 in various portions of Eastern Bay and its tributaries (see Table 32).

Table 32. Historic documentation of SAV by decades, Eastern Bay, Maryland (Area 4)

Species	1960	1970
Potamogeton perfoliatus	1961(M) 1960-1961(B) 1960-1962(H) 1968(V)	1973(H) 1971-1976(MB) 1975(G) 1976(F)(P)
Ruppia maritima	1961(M) 1960-1961(B) 1962(H) 1968(V)	1973(H) 1971-1976(MB) 1975(G) 1976(F)(P)
Myriophyllum spicatum	1961(M) 1968(V)	1976(F)(P) 1972(H) 1972-1974,1976(MB) 1975(G)
Zostera marina	1960-1961(B) 1961(M) 1968(V)	1971-1972,1975(MB) 1976(F)
P. pectinatus	1961(B)(M) 1968(V)	1972-1974,1976(MB) 1976(F)(P)
Zannichellia palustris	1961(M) 1968(V)	1972,1976(MB) 1975(G) 1976(F)(P)
Elodea canadensis	1961(B)(M) 1968(V)	1972-1974,1976(MB) 1975(G) 1976(F)(P)
Chara spp.	1961(M) 1968(V)	1975(G)
Najas spp.	1961(M)	1975(G) 1976(F)
Ceratophyllum demersum	1961(M)	1976(F)

From June 28 to August 27, 1976, Fenwick (unpublished) surveyed the Wye and Miles Rivers, Kent Island, Rich Neck and Kent Narrows Bridge to Bennett Point. *Ruppia* was the dominant species in the Miles River, *Zannichellia* dominant in the Wye River, *P. pectinatus* most common at Rich Neck and *P. perfoliatus* dominant at Kent Island. Other species documented included *Elodea*, *Najas*, *Myriophyllum*, *Vallisneria*, *Ceratophyllum* and *Zostera*.

Area 5. Choptank River

In the 1930s, four species of SAV were documented by herbaria (see Table 33). Of these four, only *Vallisneria* has not been found by recent survey teams. *Vallisneria* was last documented in 1968 at only 1 out of 146 sampling stations during the Vegetation Survey.

In surveying for waterfowl populations in the 1950s, Stewart (1962) observed what he termed to be "fair" beds of *Zostera* along with *Ruppia* and *P. pectinatus* growing in the open estuarine bays of the lower Choptank. The upper estuarine bay of the Choptank supported *Elodea*, *Ruppia* and *P. pectinatus*. Observations of submerged aquatic plants from selected locations along the Choptank River have been made by Horn Point Environmental Laboratory personnel from 1974 to present. In 1974 dense populations of *Zannichellia* were found in the marsh embayment of Horn Point (Stevenson, personal communication). In 1975, several researchers noticed severely reduced populations of submerged aquatics in the lower Choptank where they had existed previously (Jones and Krantz, personal communication). In June 1976 surveys were made of Trappe Creek and LeCompte Bay. *Zannichellia* and *Myriophyllum* were found in great abundance with a mixture of other species.

Table 33. Historic documentation of SAV by decades, Choptank River, Maryland (Area 5)

Species	1930	1940	1950	1960	1970
Potamogeton perfoliatus	1939(H)		1959(B)[a]	1961(B) 1968(V)	1970(H) 1971-1976(MB) 1977(HP)
Ruppia maritima	1939(H)		1950s(RS) 1959(B)	1961(B) 1968(V)	1971-1976(MB)
Myriophyllum spicatum				1969(V)	1976-1977(HP)
Zostera marina	1939(H)		1950s(RS)	1961(B) 1968(V)	1971-1972(MB) 1977(HP)
P. pectinatus			1950s(RS)	1969(V)	1971-1973(MB)
Zannichellia palustris				1969(V)	1972,1976(MB) 1974-1977(HP
Vallisneria americana	1933(H)			1968(V)	
Elodea canadensis		1940(H)	1950s(RS)	1961(B) 1968(V)	1971-1974,1976(MB) 1977(HP)

[a] The 1959 Benthic Survey listed *Potamogeton* sp. by genus only.

In early March (1977), just after the ice disappeared, *Myriophyllum* was identified in Trappe Creek with no other species appearing for about five weeks. The green *Myriophyllum* sprigs evidently had overwintered intact under the severe winter ice. *Zannichellia* then started becoming the dominant species in the Horn Point area (especially thick around Todd Point), primarily growing by seed germination. Commonly mixed with the *Zannichellia* were patches of *P. perfoliatus* and occasional growth of *Elodea*. The *Zannichellia* started going to seed the first weeks of June, and the *P. perfoliatus* soon after. In late June *Zostera* and *Ruppia* were found in the Todd Point area where the *Zannichellia* had suddenly died back. In Trappe Creek new growth of *Myriophyllum* and *P. perfoliatus* was observed (Cunningham, personal communication).

Area 6. Little Choptank River

Stewart (1962) surveyed the Little Choptank in the 1950s and found P. pectinatus along with unnamed salt-water species. Ruppia was found documented 97 percent of vegetation survey transects in 1968, Zostera was found at 80 percent and P. pectinatus showed up at almost 50 percent of the 65 transects sampled. Though the MBHRL Survey documented no vegetation during 1973 to 1975, Ruppia was found prior to 1973 and also in 1976 in Phillips Creek and off Smith Cove.

Table 34. Historic documentation of SAV by decades, Little Choptank River, Maryland (Area 6)

Species	1930	1950	1960	1970
Potamogeton perfoliatus			1960(B) 1968(V)	
Ruppia maritima	1931(H)		1960(B) 1968(V)	1971-1972,1976(MB)
Zostera marina			1960(B) 1968(V)	1971(MB)
P. pectinatus		1950s(RS)	1968(V)	1971-1972(MB)

Zostera was documented in 1971 at one station off McKeil Point along with P. pectinatus. The following year no vegetation was found at that station. Trace amounts of P. pectinatus were noted in 1972 in the area of Slaughter and Parsons Creeks. The same year, a substantial growth of Zannichellia was documented in Phillips Creek, associated with Ruppia.

Area 7. James Island and Honga River

Zostera and Ruppia historically have dominated the waters around James, Hooper and Taylor Island until 1972 (see Table 35). Though a decline in percentage occurrence of Ruppia was noted by MBHRL found 1971 to 1976, the species remained in the area in contrast to Zostera. This latter species was found to be abundant in 1971 at 41 percent of the 34 stations sampled. By the following year, only 3 percent of the same stations showed Zostera and no Zostera was found from 1973 to 1976.

Stewart (1962) surveyed Tar Bay in the 1950s. He recorded P. pectinatus as one of the principal species of SAV. Elser (1969) visited Tar Bay in July of 1967 and observed an extensive, dense stand of Zostera that he estimated to cover several thousand acres. Ruppia appeared to be subdominant. During the following month, Zostera appeared to succumb to a disease and slowly be replaced by Ruppia.

Table 35. Historic documentation of SAV by decades, James Island and Honga River, Maryland (Area 7)

Species	1950	1960	1970
Ruppia maritima		1960-1961(B) 1967(V) 1967(E)	1971-1976(MB)
Zostera marina		1960-1961(B) 1967(V)(E)	1971-1972(MB)
Potamogeton pectinatus	1950s(RS)	1967(V) 1968(H)	1972(MB) 1973(H)
Zannichellia palustris			1976(MB)

Abundant _P_. _pectinatus_ was observed in 1967 by the Vegetation Survey, but in 1971 MBHRL personnel found none at the 32 sampling stations. In 1972, _P_. _pectinatus_ was again documented, this time at 20 percent of the stations sampled. One herbarium specimen dated 1973 indicates that some _P_. _pectinatus_ was still present the following year.

Zannichellia was not documented this far south until 1976. Trace quantities of the species were found in Cow Cove off lower Hooper Island. Results of the 1977 MBHRL Survey may show whether or not this species has been able to become established.

Area 8. Honga River

Stewart (1962) cites _P_. _pectinatus_ as one of the principal species of SAV in the Honga River. The Vegetation Survey documented extensive growth of _Zostera_, _Ruppia_ and _P_. _pectinatus_ in the Honga River in 1967 (see Table 36). In 1971, MBHRL teams observed _Zostera_ and _Ruppia_ still in abundance but did not note any _P_. _pectinatus_. This latter species did turn up in 1972 and 1973 in "more than trace" (Kerwin et al. 1975b) quantities. However, from 1974 through 1976, the survey team again found no _P_. _pectinatus_.

From 1971 to 1976, _Ruppia_ was documented each year though at a decreasing number of stations. _Zostera_ also decreased over the same six years and seemed to disappear in 1973 and 1974; then returned in 1975 and 1976. Trace amounts of _Zannichellia_ were observed the summer of 1976.

Table 36. Historic documentation of SAV by decades, Honga River, Maryland (Area 8)

Species	1930	1950	1960	1970
Ruppia maritima			1960-1961(B) 1967(V)	1971-1976(MB)
Zostera marina	1933(H)		1960-1961(B) 1967(V)	1971-1972(MB) 1975-1976(MB)
Potamogeton pectinatus		1950s(RS)	1967(V)	1972-1973(MB)
Zannichellia palustris				1976(MB)

Area 9. Bloodsworth and South Marsh Islands

The Vegetation and Benthic surveys of the 1960s (see Table 37) documented extensive growths of *Ruppia*, *Zostera* and *P. pectinatus*. These same species have been found by the MBHRL Survey teams; however, percentage occurrence figures indicate that both *Ruppia* and *Zostera* have decreased in occurrence since 1971. It is interesting to note that *Zostera* was not found at all in the 1976 survey (see *Zostera* in Chapter 1). *P. pectinatus* has not been documented since 1973 when it was noted off Great Cove and south of Spring Island. Stotts (personal communication) noted *Zannichellia* along the midsection of South Marsh Island in June, 1976.

Table 37. Historic documentation of SAV by decades, Bloodsworth Island, Maryland (Area 9)

Species	1959	1960	1970
Ruppia maritima	1959(B)	1960-1961(B) 1967(V)	1971-1976(MB)
Zostera marina		1960-1961(B) 1967(V)	1971-1975(MB)
Potamogeton pectinatus		1967(V)	1971-1973(MB)
Zannichellia palustris			1976(VS)

Area 10. Susquehanna Flats

Submerged aquatic macrophytes have been documented on the Susquehanna Flats since the late 1870s (see Table 38). With the initiation of the Susquehanna Flats Survey in 1958, the changes in occurrence of *Myriophyllum*, *Elodea*, *Vallisneria*, and *Najas* were followed to the present. All native species have decreased in abundance since 1958; only *Myriophyllum* was found from 1971 to 1975. *Najas* was again seen in 1977 but was classified as "rare" (Stotts, personal communication). Milfoil Survey teams worked on the Flats in September of 1969 and found dense populations of *Vallisneria*. Considerable break-up of the top portions of the plants was noted in the fall. The MBHRL Survey substantiated the decline and disappearance of *Elodea*, *Najas* and *Vallisneria* and trace amounts of the latter species were documented in 1972 in Buzzard Bay.

Table 38. Historic documentation of SAV by decades, Susquehanna Flats, (Maryland (Area 10)

Species	1870-1900	1900	1950	1960	1970
Potamogeton perfoliatus	1879(H) 1895(H)	1902(H)	1953(H) 1959(S)	1961(M) 1963(H)	
Myriophyllum spicatum			1952(H) 1958-1959(S)	1950-1969(S) 1961(M) 1967-1969(M) 1962,1963(H) 1965,1968(H)	1970-1971(M) 1970-1975(S) 1971(MB) 1974-1976(MB)
P. pectinatus[a]	1879(H)	1902(H)	1953(H) 1959(S)		
Vallisneria americana		1902(H)	1951,1953(H) 1958-1959(S)	1960-1969(S) 1961(M) 1963(H)	1970-1971(S) 1971-1972(MB)
Elodea canadensis	1879(H)		1951-1953(H) 1958-1959(S)	1960-1969(S) 1963(H)	1970-1971(S) 1971(MB)
Najas spp.			1958-1959(S)	1960-1961(M) 1960-1969(S) 1963(H)	1970-1971(S) 1971(MB) 1977(S)
Chara spp.			1959(S)		1971(MB)
Ceratophyllum[a] *demersum*		1902(H)	1953(H)	1961(M) 1962,1963(H)	1971(MB)

[a] These species plus *Zannichellia palustris* were listed as non-dominant species in the Susquehanna Flats Survey. Occurrence and abundance were not stated except for the total period of 1958-1975.

Area 11. Fishing Bay

Three surveys have included sampling stations in Fishing Bay: Benthic Survey; MWA Vegetation Survey; and MBHRL Survey (see Table 39). The Benthic Survey had one transect across upper Fishing Bay in 1959 but no SAV was noted. In 1967, MWA teams found *Zostera* and *P. pectinatus* at just over 50 percent of the 27 transects sampled and *Ruppia* at 37 percent of the transects. MBHRL documented *Ruppia* and *Zostera* in 1972. *Ruppia* was found at two stations off Duck and Clay Island marshes and measured as 99 percent per volume of sample. *Zostera* was documented each year as a trace amount, in Duck Island Cove in 1971 and then off Fishing Point in 1972. After 1972 no submerged aquatic species were found by survey crews.

Table 39. Historic documentation of SAV by decades, Fishing Bay, Maryland (Area 11)

Species	1960	1970
Ruppia maritima	1967(V)	1971(MB)
Zostera marina	1967(V)	1971-1972(MB)
Potamogeton pectinatus	1967(V)	
Chara spp.		1971(H)

Area 12. Nanticoke and Wicomico Rivers

Survey teams from the Milfoil Survey and MWA documented five SAV species in the Nanticoke and Wicomico Rivers from 1965 to 1969. *Ruppia* and *P. perfoliatus* were again found in 1971 but since then, the MBHRL crews have documented no vegetation at any of the 30 stations that have been sampled annually since 1971 (see Table 40).

Table 40. Historic documentation of SAV by decades, Nanticoke and Wicomico Rivers, Maryland (Area 12)

Species	1950	1960	1970
Potamogeton perfoliatus		1965(M) 1967-1969(M)	1971(M)
Ruppia maritima		1965(M) 1967-1968(M) 1967(V)	1970-1971(M)
Myriophyllum spicatum	1957(H)	1965,1967(M)	
P. pectinatus		1969(M)	

Area 13. Manokin River

In sharp contrast to the complete absence of SAV in the nearby Nanticoke and Wicomico Rivers after 1971, the MBHRL Survey indicated Ruppia from 1971 through 1976 and Zostera from 1971 to 1975 in the Manokin River (see Table 41). However, percentage occurrence based on number of vegetated sampling stations indicates declines for both species over time. Zostera was documented at 33 percent of the 15 sampling stations in 1971, 7 percent in 1975 and zero in 1976. Ruppia's decline was less drastic; 20 percent of the stations were vegetated in 1971, 7 percent in 1975 and almost 7 percent in 1976. In 1972, P. pectinatus was documented at the mouths of Teague and Goose Creeks but has not been found since then.

Table 41. Historic documentation of SAV by decades, Manokin River, Maryland (Area 13)

Species	1960	1970
Ruppia maritima	1967(V)	1971-1976(MB)
Zostera marina	1967(V)	1971-1975(MB)
Potamogeton pectinatus		1972(MB)

Area 14. Patapsco River

Ruppia and Elodea are documented with herbarium specimens from the 1950s; however, there is no documented evidence of Elodea since then (see Table 42). Ruppia was noted as occasional in Main Creek in 1961 by the Milfoil Survey personnel and was found to be locally abundant in Back Creek. MBHRL teams have not observed Ruppia or Elodea in the Patapsco River during their six years of sampling.

In 1961, Myriophyllum was found to be locally abundant in Main Creek and occasional to rare in Back, Rock and Fox Creeks. P. perfoliatus was considered to be generally rare, though locally abundant in Back Creek. Vallisneria, the only other SAV species observed in 1961, was locally abundant in Back Creek and rare to occasional in all other creeks that were sampled.

There is no information indicating any sampling of SAV in the Patapsco River from 1962 to 1971 except for one herbarium specimen for Najas dated 1968. The MBHRL Survey teams have documented three species from 1972 to 1976, the two Potamogeton spp. and Vallisneria. However, data indicate only trace amounts of all three species except for Vallisneria in 1974, when analysis was indicated as 99 percent per volume from two samples near the mouth of Stony Creek.

Table 42. Historic documentation of SAV by decades, Patapsco River, Maryland (Area 14)

Species	1950	1960	1970[a]
Potamogeton perfoliatus		1961(M)	1972-1974(MB)
Ruppia maritima	1953(H)	1961(M)	
Myriophyllum spicatum		1961(M)	
P. pectinatus			1972(MB)
Vallisneria americana		1961(M)	1973-1974(MB) 1976(MB)
Elodea canadensis	1950(H)		
Najas spp.		1968(H)	

[a] MBHRL did not sample the Patapsco River in 1975

Area 15. Big and Little Annemessex Rivers

Ruppia and Zostera are documented as the dominant species in the Annemessex Rivers (see Table 43). Both species were documented from 1959 to 1961 during the Benthic Survey and have been found consistently over the six years of the MBHRL Survey. However, percentage occurrence of sampling stations indicates that both species have declined since 1971; Ruppia was found at 45 percent of the 20 stations in 1971 and at 25 percent in 1976; and Zostera declined from 60 percent in 1971 to 15 percent in 1976.

Table 43. Historic documentation of SAV by decades, Big and Little Annemessex Rivers, Maryland (Area 15)

Species	1960	1970
Ruppia maritima	1959-1961(B) 1967(V)	1971-1976(MB)
Zostera marina	1959-1961(B) 1967(V)	1971-1976(MB)
Potamogeton perfoliatus	1967(V)	1971-1973(MB)

Potamogeton pectinatus is the only other species whose presence has been documented in this area. During the 1967 Vegetation Survey, 69 percent of the stations sampled in the Big Annemessex River were vegetated with *P. pectinatus*. Trace amounts were noted in MBHRL analyses of 1971 and 1973 samples from the Little Annemessex. In 1972, the species was found in "greater than trace" (Kerwin et al. 1975*b*) amounts at several stations in the Big Annemessex in Fords and Crane Coves.

Area 16. Gunpowder and Bush Rivers

Historic records for the Gunpowder and Bush Rivers date back to herbarium specimens of 1895 for *Myriophyllum*, *P. perfoliatus* and *P. pectinatus* (see Table 44). There is no indication that *P. pectinatus* and *Ruppia* (documented in 1902 and 1903 only) were able to successfully persist. Milfoil Surveys and herbarium specimens documented a total of eight SAV species in this area during the 1960s. In 1970 and 1971, five of these same species were found. However, since 1971, MBHRL has documented no vegetation in the headwaters of the two rivers. MBHRL has not sampled the southern portions of the rivers.

Elser (1969) surveyed the Bird River in 1965 and found large beds of *Myriophyllum*, *Vallisneria* and *Ceratophyllum*. When milfoil retreated the following year, *Vallisneria* and *Ceratophyllum* colonized the vacated areas. However, in 1967, the same areas were noted to be nearly bare with only a few, sparse beds.

Table 44. Historic documentation of SAV by decades, Gunpowder and Bush Rivers, Maryland (Area 16)

Species	1890	1900	1950	1960	1970
Potamogeton perfoliatus	1895(H)	1902(H)		1961(H)(M)	1970-1971(M)
Ruppia maritima		1902(H) 1903(H)			
Myriophyllum spicatum	1895(H)			1960,1961(H) 1962,1963(H) 1964-1967(E) 1961(M) 1967-1969(M)	1970(H) 1971(MB) 1970-1971(M)
P. pectinatus	1895(H)				
Zannichellia palustris				1961(M)	
Vallisneria americana		1902(H)	1951(H) 1956(H)	1961,1963(H) 1965-1967(E) 1961,1969(M)	1970-1971(M)
Elodea canadensis				1961(M)	1971(M)
Najas spp.			1956(H)	1961(M, 1963(H)	1970-1971(M)
Chara spp.				1961(M)	
Ceratophyllum demersum			1956(H)	1961(H) 1965-1967(E)	1970-1971(M)

When Milfoil Survey teams looked at Gunpowder River in 1969, 1970 and 1971, *Myriophyllum* was found to be dominant south of the railroad bridge. *Vallisneria* and *P. perfoliatus* were found along with traces of *Najas* and *Ceratophyllum*. In Dundee and Saltpeter Creeks, the water was observed to be only slightly turbid in May and June of 1960. Shoal areas were covered with extensive diversified growth. The excessive rains and runoff in July seemed to affect the vegetation in Saltpeter Creek resulting in almost complete loss. Dundee Creek was not as affected and extensive beds of *Myriophyllum* and *Vallisneria* were noted. In July of 1977, MBHRL Survey teams found abundant *Vallisneria* and *Chara* in Dundee Creek. *Myriophyllum* and *P. crispus* were observed growing in water over 2 m deep.

Potamogeton foliosus, a species that had not been seen for about 10 years, was noted in the Gunpowder River in 1971. The MBHRL Survey did not indicate the continued presence of this species subsequent to the scattered growth that was noted by the Milfoil team.

Area 17. Pocomoke Sound

The Vegetation Survey sampled 19 transects in Pocomoke Sound in 1967 and documented *Zostera* at 89 percent and *Ruppia* at 58 percent of the stations (see Table 45). Except for 1974, MBHRL Survey teams have sampled from 20 to 22 stations each year in the Maryland portion of Pocomoke Sound. As would be expected in the high salinity waters of the Sound, *Ruppia* and *Zostera* were the only species documented. Based on percentage occurrence figures, both species show declines over the six year MBHRL Survey. *Zostera* was observed at 18 percent of the stations sampled in 1971 compared to 9 percent in 1976. *Ruppia* declined from 9 percent to 4.5 percent over the same six years and was noted at all in 1972 and 1973.

Table 45. Historic documentation of SAV by decades Pocomoke Sound, Maryland (Area 17)

Species	1960	1970[a]
Ruppia maritima	1967(V)	1971,1975-1976(MB)
Zostera marina	1967(V)	1971-1973(MB) 1975-1976(MB)

[a] MBHRL did not sample Pocomoke Sound in 1974

Area 18. Magothy River

In the Magothy River, Milfoil Survey teams, Vernon Stotts, Harold Elser and MBHRL have documented nine species of SAV over the 17-year period from 1960 to 1976 (see Table 46). A saltwater intrusion noted prior to 1967 and allowed SAV species other than *Myriophyllum* to become reestablished (Elser 1969). *Ruppia* and *P. perfoliatus* were found in 1969 to be dominant in such areas as the shoals off Broad Creek, in Deep Mill Creek and in the upper reaches of Blackhole Creek. In the embayments beyond the confluence of Cockey, Magothy, and Cattail Creeks, the water was very turbid in August, 1969, and almost no SAV was noted by the Milfoil Survey team personnel.

P. perfoliatus was documented consistently until 1974 when it was last observed at two locations off Park and Grays Points. *Ruppia*, *Chara* and *Najas* have not been documented since 1974 despite annual sampling efforts by MBHRL. Only *Vallisneria* has been noted in the Magothy River since 1974 and 1976 when only a trace amount was noted off Pea Patch Point.

Table 46. Historic documentation of SAV by decades, Magothy River, Maryland (Area 18)

Species	1900	1940	1950	1960	1970
Potamogeton perfoliatus		1945(H)	1950(H) 1959(B)[a]	1960-1961(B) 1961,1969(M) 1969(E)	1971(MB) 1973-1974(MB) 1970-1971(M)
Ruppia maritima		1945(H)	1950(H) 1959(B)	1960-1961(B) 1961,1969(M)	1971,1974(MB) 1970-1971(M)
Myriophyllum spicatum				1961(M) 1968-1969(M) 1967(E)	1970(H) 1970-1971(M)
P. pectinatus				1960(B) 1961,1969(M)	1970-1971(M) 1971,1973(MB)
Zannichellia palustris					1970-1971(M)
Vallisneria americana	1902(H)			1960-1961(B) 1961(M)	1970-1971(M) 1971,1973(MB) 1976(MB)
Elodea canadensis				1961(M)	
Najas spp.				1960-1961(B) 1961(M)	1971(MB) 1973-1974(MB)
Chara spp.				1961(B)(M)	1974(MB)
Ceratophyllum demersum		1947(H)			

[a] The 1959 Benthic Survey listed *Potamogeton* sp. by genus only

Area 19. Severn River

Submerged aquatic species have been documented in the Severn River since 1893 (see Table 47). In the 1950s and 1960s, Milfoil Survey and herbarium records show 8 species present. Survey teams found abundant *Elodea* in July, 1961, along the south shore of the river. Elser (1969) found abundant *P. perfoliatus* and *Myriophyllum* prior to August, 1966. By 1967, Elser noted only a few, sparse beds of the same species. In September, 1969, the waters of the Servern River were found to be cloudy in most embayments and indentations. In Asquith Creek and Ringold Cove abundant growth of *P. perfoliatus* was observed along with scattered *Ruppia* and *P. pectinatus*. Across the river, SAV seemed to be practically nonexistent. In the Severna Park area and across the river in creek embayments above Long Point, scattered growth of *P. perfoliatus* and *Ruppia* were commonly found. By 1969, no vegetation was observed in the upper reaches of the river. *P. perfoliatus* appeared to be the dominant species in the 1970s; however, between 1971 and 1976 percentage occurrence figures indicated a decline of about one-half in the number of vegetated stations.

Table 47. Historic documentation of SAV by decades, Severn River, Maryland (Area 19)

Species	1890	1920	1940	1950	1960	1970
Potamogeton perfoliatus		1927(H)		1952(H)	1961,1969(M) 1966-1967(E)	1970-1971(M) 1976(MB) 1971-1974(MB)
Ruppia maritima		1927(H)		1952(H)	1961,1969(M)	1970-1971(M) 1971-1974(MB) 1976(MB)
Myriophyllum spicatum					1961(M) 1966-1967(E) 1968-1969(M)	1970(H) 1970-1971(M) 1972,1976(MB)
P. pectinatus	1893(H)	1927(H)		1952(H)	1961,1969(M)	1970-1971(M) 1972-1974(MB)
Zannichellia palustris						1970-1971(M) 1974,1976(MB)
Vallisneria americana					1961(M)	
Elodea canadensis			1947(H)	1952(H)	1961(M)	
Najas spp.				1954(H)	1961(M) 1969(H)	1970-1971(M) 1971-1974(MB)
Chara spp.					1961(M)	1972-1974(MB) 1976(MB)
Ceratophyllum demersum			1949(H)	1952(H)		1974(MB)

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[a] MBHRL did not sample the Severn River in 1975

Area 20. Patuxent River

From the 1930s to present, eight species of SAV were documented in the Patuxent River (see Table 48). Elser (1969) observed healthy *Zostera* at Solomons in the late 1940s. Stewart (1962) listed *P. pectinatus* plus unnamed salt-water species as present in the lower Patuxent River from the mouth upstream to Point Patience. Herbarium specimens plus the sampling work of the Milfoil Survey and Vernon Stotts provide evidence of occurrence in the 1960s. In July, 1961, Milfoil teams sampled the west side of the Patuxent River south of the Benedict Bridge and noted *Elodea* to be generally abundant in most of the creeks. *Ruppia* and *P. perfoliatus* ranged from rare to abundant, *P. pectinatus* was observed to be locally abundant in St. Thomas and Cuckold Creeks and *Zannichellia* was listed as rare in Mill and Washington Creeks.

Table 48. Historic documentation of SAV by decades, Patuxent River. Maryland (Area 20)

Species	1930	1940	1950	1960	1970
Potamogeton perfoliatus			1952(H)	1961(M) 1961,1964(H) 1966(H) 1963-1965(A)	1970-1971(M)
Ruppia maritima	1937(H)	1946(H)		1961(M) 1963-1965(A)	1970-1971(M) 1972,1974(MB) 1976(MB)
Zostera marina	1933,1937(H)	1940s(E) 1946(H)		1961(B) 1965(H)	1971(MB)
P. pectinatus	1937(H)	1946(H)	1950s(RC)	1961(M)	1970-1971(M)
Zannichellia palustris				1961(M) 1965(H)	
Elodea canadensis			1954(H)	1961(M) 1965(H)	
Najas spp.	1939(H)	1947(H)	1954(H) 1957(H)	1969(H)	
Vallisneria americana				1965(H)	

The east side of the river near Solomons was also sampled in July, 1961. Vegetation was limited to *Ruppia* and was found to be generally sparse. MBHRL and Milfoil surveys from 1970 to 1976 documented only three of the eight species that had been observed during the previous decade. A trace of *Zostera* was noted in Cuckold Creek in 1971. Milfoil Survey teams described a general lack of extensive vegetation in the entire river and only sparse growths of *Ruppia*, *P. pectinatus* and *P. perfoliatus* in 1970 and 1971. Trace amounts of *Ruppia* were documented in 1972 off Sheridan Point and in Cuckold Creek. Only *Ruppia* was found again in 1974 and 1976 at the Cukold Creek station.

Area 21. Back, Middle and Gunpowder Rivers

Submerged aquatic species diversity was high in the 1960s in these three rivers; ten species were documented in 1960, 1961 and 1969 (see Table 49).

By 1970, diversity had decreased to six species with the loss of Najas, P. pectinatus, Chara and Zannichellia. Good growths of Vallisneria were seen in Galloway Creek in 1969 with lesser amounts of P. perfoliatus and traces of Zannichellia, P. pectinatus, Ruppia and Najas. The following year, scattered patches of Vallisneria and Ruppia and some of the other pondweeds were noted throughout Middle River.

Table 49. Historic documentation of SAV by decades, Back, Middle and Gunpowder Rivers, Maryland (Area 21)

Species	1940	1960	1970
Potamogeton perfoliatus	1940(H)	1960-1961,1969(M)	1970-1971(M)
Ruppia maritima		1961,1969(M)	1970(M) 1971(MB)
Myriophyllum spicatum		1960-1961(B) 1960-1961,1969(M) 1962,1963(H) 1967-1969(SO) 1968(H)	1970-1971(M) 1971-1973,1975(MB)
P. pectinatus		1960-1961,1969(M) 1963(H)	
Zannichellia palustris		1969(M)	
Vallisneria americana	1940(H)	1960-1961(B) 1960-1961,1969(M)	1970-1971(M) 1974-1975(MB)
Elodea canadensis	1940(H)	1960-1961,1969(M)	1970-1971(M) 1972(MB)
Najas spp.		1960-1961,1969(M) 1961(B)	
Chara spp.		1960(M)	
Ceratophyllum demersum	1940(H)	1960-1961,1969(M)	1970-1971(M)

Back River was reported to be very turbid in 1969 and no submerged vegetation was documented. Vallisneria showed up in 1974 and 1975 but not in 1976 at the mouth of Seneca Creek. Ruppia was documented in the same area in Seneca Creek by MBHRL in 1971 but has not been found since. Along with Vallisneria, Myriophyllum is the only other SAV species that was seen in 1975. The 1976 MBHRL Survey found no vegetation in Area 21.

Area 22. Curtis and Cove Points

Herbarium specimens and Benthic Survey transects taken along the western Bay shore between Curtis and Cove Points have documented four SAV species between 1931 and 1965 (see Table 50). Despite about 20 stations sampled from 1971 to 1976, the MBHRL teams found no vegetation. Although the high energy nature of this portion of the Chesapeake Bay is not conducive to SAV invasion and growth, it is interesting to note that there are past records of species occurrence in these areas.

Table 50. Historic documentation of SAV by decades, Curtis and Cove Points, Maryland (Area 22)

Species	1930	1940	1960
Potamogeton perfoliatus			1963(H)
Ruppia maritima		1948(H)	1961(B)
Zostera marina			1965(H)
Zannichellia palustris	1931(H)		

Area 23. South, West and Rhode Rivers

Elser (1969, p. 58) reported that the Rhode River supported"....rather heavy weed load..." in 1964. Myriophyllum was dominant but the river also included large beds of P. perfoliatus and lesser amounts of Elodea, P. pectinatus, Ruppia and Zannichellia were also represented (see Table 51). In 1965, Elser observed a general decline of vegetation over the summer. By 1966, SAV coverage appeared to be about 10 percent of the 1965 observed abundance. By 1967, Elser found only a few sparse beds of P. perfoliatus. This decline Elser referred to as a phenomenon and named it the "Rhode River Evanescence."

Southwick and Pine (1975) studied the complete shoreline of the Rhode River and its tributaries from 1966 to 1973. Based on their observations, they concluded that the six species studied had declined substantially and Elodea had disappeared completely after 1966. Those species that were observed until 1973 were Ruppia, P. perfoliatus, Myriophyllum and Zannichellia. In 1961, the Milfoil Survey teams surveyed the Rhode River and observed P. perfoliatus growing abundantly in most of the creeks. Elodea and Ruppia were considered as occasional to common inhabitants of the same area. A 1945 herbarium specimen places Vallisneria in the Rhode River but subsequent survey work has not documented the species since. In 1969, P. perfoliatus was still observed to be abundant. Lesser amounts of P. pectinatus, Ruppia and a trace of Myriophyllum were noted prior to a mid-July die-back. MBHRL Survey teams did not find any vegetation from 1971 to 1976.

Milfoil teams surveyed West River in August of 1961 and found P. perfoliatus to be growing abundantly in most of the sheltered areas of the river. Elodea and Ruppia were considered common in most creeks and Myriophyllum and P. pectinatus turned up occasionally. Ruppia showed up in West River in 1976, for the first time since MBHRL started surveying in 1971.

Table 51. Historic documentation of SAV by decades, South, West and Rhode Rivers, Maryland (Area 23)

Species	1900	1940	1950	1960	1970
Potamogeton perfoliatus				1960,1963(H) 1961,1969(M) 1962(P&B) 1964-1967(E) 1966-1969(S&P)	1970-1971(M) 1972-1973(S&P) 1976(CB)
Ruppia maritima	1905(H)	1946(H)	1953(H)	1961,1969(M) 1962(P&B) 1963(H) 1966-1969(S&P)	1970-1971(M) 1971-1973(S&P) 1976(MB) 1976(CB)
Myriophyllum spicatum				1961,1968(M) 1962(P&B) 1964-1967(E) 1966-1969(S&P)	1970(M) 1970-1973(S&P)
P. pectinatus		1947(H)		1961,1969(M) 1964-1967(E) 1966-1969(S&P)	1971-1973(S&P) 1976(CB)
Zannichellia palustris				1961(M) 1966-1969(S&P)	1970-1973(S&P) 1976(CB)
Vallisneria americana		1945(H)			
Elodea canadensis		1945(H)		1960(H) 1961(M) 1962(P&B) 1966(S&P)	

Phillipp and Brown (1965) surveyed two locations in the South River in 1962 and found dense masses of *P. perfoliatus* and *Elodea nuttallii* ringing Limehouse Cove and from Mayo Point to Long Point. Milfoil teams found *P. perfoliatus* to be abundant in 1961 in Ramsay Lake, Selby Bay and Duvall Creek and common to occasional growth in most of the remainder of South River. *Elodea*, *Myriophyllum* and *Ruppia* were determined to be generally common to the area. MBHRL teams did not observe any vegetation in the South River from 1971 to 1976.

Personnel from the Cheapeake Bay Center for Environmental Studies (CBCES) surveyed the Rhode River in June and July, 1976. *Ruppia* was determined to be dominant, along with abundant *Zannichellia*. Traces of *P. perfoliatus* and *P. pectinatus* were also found.

Area 24. Chester River

Through the combined efforts of the Milfoil teams, MWA and MBHRL, aquatic species occurrence and abundance in the Chester River have been well documented from 1960 to 1976 (see Table 52). Results from the Benthic Survey (1959 to 1961) show *P. perfoliatus* as dominant and abundant at virtually all stations sampled. *Najas* and *Myriophyllum* were found frequently and *P. pectinatus* and *Elodea* found occasionally. During 1964 through 1966, increased salinity pushed *Myriophyllum* further up river and allowed P. perfoliatus to reestablish. In

1965, Milfoil personnel sampled much of the Chester River and found P. perfoliatus, Vallisneria, Myriophyllum and Ruppia locally dominant and abundant. Elodea was documented occasionally. Excessive precipitation and turbidity from runoff resulted in heavy clouded water throughout the growing season in 1967 and in mid-July and early August of 1969. In 1968, scattered growth of P. perfoliatus was observed in the Chester River. By 1969, the Vegetation Survey cited P. perfoliatus as dominant and abundant. Ruppia and Myriophyllum were found at over 50 percent of the transects sampled. Vallisneria which had been found so abundant in 1968 was observed at less than 25 percent of the transects. Chara and Najas were observed at over 25 percent of the transects.

In the 1970s survey teams documented 10 of the 11 previously seen species in the Chester River. P. perfoliatus remained dominant through 1976. Myriophyllum and Ceratophyllum were the only species not found during the 1975 and 1976 surveys. Trace amounts of Ceratophyllum were seen in 1970 in various Chester River embayments but these growths were virtually gone in 1971. Based on the frequency of occurrence data from MBHRL, total vegetation for the Chester River declined from 1971 to 1976. In 1971, 60 percent of the 36 sampling stations were vegetated as compared to 1976 when 25 percent showed vegetation.

Table 52. Historic documentation of SAV decades, Chester River, Maryland (Area 24)

Species	1900	1940	1950	1960	1970
Potamogeton perfoliatus		1947(H)	1959(B)[a]	1961(M) 1965(H) 1965,1968-1969(M) 1969(V)	1970-1971(M) 1971-1976(MB) 1972(H)
Ruppia maritima		1947(H)	1959(B)	1960-1961(B) 1961,1965(M) 1965(H) 1969(V)	1970-1972(P) 1970-1971(M) 1971-1976(MB)
Myriophyllum spicatum				1961(B)(M) 1965,1968-1969(M) 1969(V)	1970-1971(M) 1971-1973(MB)
Zostera marina				1969(V)	
P. pectinatus	1907(H)			1960-1961(B) 1961(M) 1969(V)	1970-1972(P) 1970-1971(M) 1971-1974,1976(MB)
Zannichellia palustris				1969(V)	1970-1972(P) 1971(M) 1973,1976(MB)
Vallisneria americana		1947(H)		1961,1969(M) 1965(M) 1968(V)	1971,1973,1976(MB)
Elodea canadensis				1960-1961(B) 1961,1965(M) 1969(V)	1970-1971(M) 1971-1972(MB) 1975-1976(MB)
Najas spp.		1947(H)		1961(B)(M) 1969(V)	1970(M) 1970-1972(P) 1971-1976(MB)
Chara spp.				1961(M) 1969(V)	1972-1975(MB)
Ceratophyllum demersum				1961(B) 1961,1965(M)	1970-1971(M)

[a] The 1959 Benthic Survey listed Potamogeton sp. by genus only.

Frank Pine, then a graduate student at The Johns Hopkins University, surveyed various areas in the Chester River in the 1970s. With Robert Munro, Pine surveyed Eastern Neck Island in 1970 and observed extensive beds of Najas, Ruppia, P. pectinatus and Zannichellia along the western shore. Pockets of the same species were noted on the eastern side of the island. By the spring of 1972, the large western beds were reduced to virtually nothing while vegetation pockets remained on the eastern side. Excessive turbidity was noted at the time.

Area 25. Love and Kent Points

The Chesapeake Bay shore between Kent and Love Points was surveyed in 1961 by Milfoil Survey teams (see Table 53). One transect was run perpendicular to the Bay shore line as part of the Benthic Survey in 1961. Both surveys documented a total of four species. By the 1969 Vegetation Survey, three species were documented with Ruppia dominant. In 1974, Ruppia alone was documented off Chews Point, north of Bloody Point, by the MBHRL Survey.

Table 53. Historic documentation of SAV by decades, Love and Kent Points, Maryland (Area 25)

Species	1960	1970
Potamogeton perfoliatus	1961(B)(M) 1969(V)	
Ruppia maritima	1961(B)(M) 1969(V)	1974(MB)
Zostera marina	1961(M) 1969(V)	
Vallisneria americana	1961(M)	

Area 26. Smith Island

Vegetation around Smith Island has been predominantly Zostera and Ruppia (see Table 54). Both species were documented by the Vegetation Survey in 1967 at the same percentage of transects though by the initation of the MBHRL Survey in 1971, Ruppia seemed to be more common than Zostera. Since that time, Zostera has declined sharply, based on percentage occurrence. Ruppia has also declined but showed a recovery in 1976. P. pectinatus was observed in both 1972 and 1973 in "more than trace" amounts. Subsequent MBHRL surveys in 1974 and 1976 have not documented any further P. pectinatus.

Table 54. Historic documentation of SAV by decades, Smith Island, Maryland (Area 26)

Species	1960	1970
Potamogeton pectinatus		1972-1973(MB)
Ruppia maritima	1967(V)	1971-1976(MB)
Zostera marina	1967(V)	1971-1972(MB) 1975-1976(MB)

Area 29. Upper Potomac River

Francis Uhler, MBHRL, is often quoted concerning his report of seeing ducks feeding pondweeds, Najas and Vallisneria, near the 14th Street Bridge in the early 1930s (see Table 55). Stewart (1962) reported finding P. pectinatus along with Najas and Vallisneria in the 1950s (date not specified). In 1952, Bartsch (1954) surveyed the upper Potomac River and determined that vegetation was essentially nonexistent. Then, from 1958 until its dramatic disappearance around 1965, Myriophyllum created nuisance conditions in the upper Potomac.

Table 55. Historic documentation of SAV by decades, Upper Potomac River, Maryland and Virginia, (Area 29)

Species	1890-1900	1900	1910	1920	1930	1940	1950	1960	1970
Potamogeton perfoliatus	1877(H) 1884,1898(H)	1906(H) 1903(H)	1914(H)	1922(H)	1930(H) 1933(H)				
Ruppia maritima			1915(H)						
Myriophyllum spicatum			1915(H)				1951(H) 1957(H)	1963(H) 1969(H)	
P. pectinatus		1903(H) 1906(H) 1908(H) 1909(H)	1915(H)		1935(H)		1952(H) 1950s(RS)		
Vallisneria americana	1889(H)	1903(H)	1914(H) 1915(H)	1925(H)	1933(H) 1930s(U)		1950s(RS)		
Elodea canadensis	1875(H) 1884,1895(H)	1903(H)	1914(H) 1918(H)		1935(H)	1941(H) 1944(H)	1950(H) 1951(H)		1977(EC)
Najas sp.				1929(H)	1933(H) 1934(H) 1939(H) 1930s(U)		1950s(RS)		1972(H)
Ceratophyllum demersum	1881,1895(H)	1903(H)	1915(H) 1918(H)		1935(H) 1936(H)	1944(H)			

MBHRL sampled the upper Potomac in 1972 but found no SAV. Based on the lack of vegetation, sampling of the entire river was dropped from the MBHRL Survey until 1977. Sampling of the same stations again documented no vegetation during the summer of 1977.

Ecological Analysts (Towson, Maryland) is presently studying Piscataway Creek for the Washington Suburban Sanitary Commission. Field personnel noted in 1977 moderate sized beds of healthy *Elodea canadensis* and patchy but healthy beds of *Potamogeton crispus* in creek bottoms just south of the mouth of Piscataway Creek in the Potomac River (Pine, personal communication).

Area 30. Upper Middle Potomac River

Port Tobacco River was surveyed from 1960 to 1971 for the Milfoil Survey. In 1969, with the exception of Goose Bay, the river was extremely turbid with only traces of *Ruppia*, *Myriophyllum* and *P. perfoliatus*. The water was clear in Goose Bay and abundant with *Myriophyllum*. By 1970, an increase in SAV was noted throughout the lower half of Port Tobacco River, principally *Vallisneria*, *Myriophyllum*, *P. perfoliatus* and *Ruppia*. In 1971, sparse SAV growth extended north to the upper marina area (see Table 56).

Table 56. Historic documentation of SAV by decades, Upper Middle Potomac River, Maryland and Virginia

Species	1910	1920	1930	1940	1950	1960	1970
Potamogeton perfoliatus	1915(H)	1921(H)	1936(H) 1938(H)	1945(H)	1954(H)	1960,1965(M) 1969(M)	1970-1971(M) 1972,1977(MB)
Ruppia maritima	1915(H)			1943(H)		1969(M)	1970-1971(M) 1972(MB)
Myriophyllum spicatum					1952(H)	1960,1965(M) 1969(M)(H)	1970-1971(M) 1972,1977(MB)
P. pectinatus			1934(H)			1969(M)	1970-1971(M)
Vallisneria americana		1927(U)	1933(H) 1934(H)	1946(H)			1970-1971(M) 1972,1977(MB)
Elodea canadensis						1969(M)	1977(MB)
Najas spp.			1935(H)				
Ceratophyllum demersum							1977(MB)

Nanjemoy Creek water was cloudy throughout 1969. Prior to the heavy rains in July, scattered *Myriophyllum* and traces of *P. pectinatus* were observed. Water in the creek remained turbid throughout 1970 and 1971 and no SAV was documented. In 1972, MBHRL sampled the Nanjemoy Creek and noted a pocket of *Vallisneria* off Blossom Point at the mouth of the creek plus some *Myriophyllum* further up on the western shore. Other than a pocket of *Vallisneria* off Taylor Neck at Riverside and a trace of the same species across the river from Quantico, no other vegetation was documented from Port Tobacco to Quantico. Sampling was not conducted by MBHRL from 1973 through 1976, but was resumed in 1977. Haramis (1977) surveyed the Potomac River for the MBHRL Survey and found that vegetation appeared to be concentrated within a 46 km area extending from Sandy Point (just south of Quantico on the Maryland shore) downstream to lower Cedar Point (just south of the 301 bridge). The majority of this river stretch is in Area 30. Haramis found lush growths of Vallisneria in Goose Bay and Port Tobacco River and P. perfoliatus and scattered Myriophyllum sp., C. demersum, E. canadensis, and P. pectinatus throughout the 46 km area. It is within this portion of the Potomac River that the fresh and salt waters mix.

Area 31. Lower Middle Potomac River

The Milfoil Survey teams observed abundant P. perfoliatus and P. pectinatus in the Breton and St. Clements Bays area in November of 1961. Myriophyllum was found locally abundant in several of the feeder creeks and Ruppia and Zostera were determined to be generally rare (see Table 57).

The Wicomico River system was sampled by Rawls in August, 1965. Myriophyllum was found in abundance in many of the creeks and bays. P. perfoliatus was abundant in St. Catherine Sound, Chaptico Bay and several creeks. Ruppia was observed occasionally and Najas was found in Dolly Boarman's Creek.

In August, 1967, Chaptico Bay and the Wicomico were found by Rawls to be quite turbid. Najas was abundant in Chaptico Bay, Ruppia occasional and P. perfoliatus common. The abundant Najas found in Dolly Boarman's Creek in 1965 was listed as rare to occasional near the mouth of the creek and occasional inside the creek. In September, 1968, Chaptico Bay was sampled again and Ruppia, Najas and P. perfoliatus were documented as occasional to abundant. Approximately 25 to 30 ha of Myriophyllum were noted. In a letter to John Steenis, dated September 25, 1968, Rawls related the status of Myriophyllum in 1968 to the explosive growth in the late 1950s and early 1960s.

Water in the Wicomico by 1970 and 1971 was progressively more turbid upstream probably due to stream channelization that was in progress at Gilbert Swamp. SAV growth was observed to be scattered with one pocket remaining at the mouth of Chaptico Bay in the Bassfold embayment. Currioman Bay and Nomini Creek were sampled from 1968 to 1971 as part of the Milfoil Survey but the waters were determined to be too turbid for extensive SAV growth. Limited growth of Ruppia and Myriophyllum plus traces of P. perfoliatus were documented. In 1972, MBHRL found a trace of Zannichellia up Nomini Creek.

In the Lower Machodoc and Glebe Creek area, no SAV was documented. Glebe Creek was observed to be very turbid from 1969 to 1971 due to extensive marina, suburban and recreational development along the shoreline. MBHRL summer sampling in 1972 documented a bed of Vallisneria just off Persimmon Point. Later, in 1976 and 1977, Uhler reported seeing sparse beds of Vallisneria at the same location. MBHRL did not sample the Potomac River after 1972 until 1977 when Haramis (1977) reported having found an extensive bed of P. perfoliatus and R. maritima on the southern shore of the 301 bridge.

Table 57. Historic documentation of SAV by decades, Lower Middle Potomac River, Maryland and Virginia (Area 31)

Species	1930	1940	1960	1970
Potamogeton perfoliatus			1961,1965(M) 1967,1969(M) 1966,1968(H)	1970(H) 1970-1971(M) 1972,1977(MB)
Ruppia maritima	1933(H) 1935(H)		1961,1965(M) 1967,1968(M) 1966,1968(H)	1977(MB)
Myriophyllum spicatum			1960,1963(H) 1961,1965(M) 1967-1969(M) 1966(H)	1970(H) 1970-1971(M)
Zostera marina			1961,1965(M) 1968(M)	1972(MB)
P. pectinatus	1933(H) 1936(H)		1961,1969(M)	
Zannichellia palustris				1970-1971(M) 1972(MB)
Vallisneria americana	1933(H)			1970,1971(M) 1972(MB) 1976-1977(U)
Elodea canadensis				1970-1971(M) 1972(MB)
Najas spp.		1946(H)	1965(M) 1967-1968(M)	1970-1971(M) 1972(MB)

Area 32. Lower Potomac River

The Milfoil Survey in 1961 documented virtually no SAV along the northern shore of the Potomac River from Point Lookout to the western point of the St. Mary's River (see Table 58).

Table 58. Historic documentation of SAV by decades, Lower Potomac River, Maryland and Virginia (Area 32)

Species	1890	1920	1960	1970
Potamogeton perfoliatus		1928(H)	1961(M)	
Ruppia maritima	1894(H)	1928(H)	1961-1965(M)	
Myriophyllum spicatum			1961(M)	
Zostera marina			1961,1963-1965(M) 1969(H)	
P. pectinatus			1961,1963(M)	1973(H)

St. George Creek, by contrast, was found to contain abundant growths of *P. perfoliatus*, *P. pectinatus* and *Ruppia*. Herring Creek was abundant with *P. perfoliatus*, *Ruppia* and *Myriophyllum*. Big Ducks Creek, just to the north, contained abundant *Myriophyllum* and occasional *Ruppia*. Blake Creek was also documented as being abundant with *Myriophyluum*.

Both shores were sampled in 1972 by MBHRL but no vegetation was found. When the MBHRL Survey was resumed in 1977, vegetation was listed as very sparse (Haramis 1977).

Rappahannock River, Virginia

The only documentation readily available for the Rappahannock River is for the 1970s. Orth studied aerial surveys of the river in order to map SAV beds but it was not possible to separate *Zostera marina* and *Ruppia martima* on on the photographs. It is probable that both species were in the river. Priest surveyed the river in the summer of 1977 and found the two species already mentioned along with *Z. palustris* and an unknown species of *Potamogeton* (see Table 59).

Table 59. Historic documentation of SAV by decades, Rappahannock River, Virginia

Species	1970
Ruppia maritima	1971-1974(0)[a] 1977(WP)
Zostera marina	1971-1974(0)[a] 1973(0)[a]
Zannichellia palustris	1977(WP)
Potamogeton (*? epihydrus*)	1977(WP)

[a] Aerial survey by Orth did not differentiate between *Zostera* and *Ruppia*

Piankatank River, Virginia

Documentation information for the Piankatank River is available only for 1971 through 1977. Orth documented *Zostera* and *Ruppia* and Priest found two beds of *Zannichellia* (see Table 60). Data is insufficient for conclusions regarding population or density changes.

Table 60. Historic documentation of SAV by decades, Piankatank River, Virginia

Species	1970
Zostera marina	1973(0) 1971, 1974(0)[a]
Ruppia maritima	1971, 1974(0)[a]
Zannichellia pulustris	1977(WP)

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[a] Aerial survey by Orth did not differentiate between Zostera and Ruppia

Mobjack Bay, Virginia

Ruppia maritima and Zostera marina are the only two species of SAV that have been documented in Mobjack Bay. Zostera was found in 1969 and also by two survey crews in the 1970s. Ruppia was sited by Orth and also by the MBHRL Survey personnel during their one-and-only survey of Mobjack Bay (see Table 61).

Table 61. Historic documentation of SAV by decades, Mobjack Bay, Virginia

Species	1960	1970
Ruppia maritima	1969 (H)	1972(MB) 1971,1974(0)[a]
Zostera marina		1972(MB) 1971,1974(0)[a]

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[a] aerial survey by Orth did not differentiate between Zostera and Ruppia

York River, Virginia

Historic documentation of aquatic vegetation in the York River spans six decades. *Zostera* has been documented fairly steadily since the 1930s (see Table 62). *Vallisneria*, *Elodea*, *Ruppia* and *P. pectinatus* have each been listed once. The lack of comprehensive survey work in Virginia waters precludes any conclusions based on these random sitings.

Table 62. Historic documentation of SAV by decades, York River, Virginia

Species	1920	1930	1950	1960	1970
Potamogeton pectinatus			1958(H)		
Ruppia maritima					1971,1974(0)[a]
Zostera marina		1937(0)	1953(0)	1960(0) 1963(0)	1970,1973(0) 1971,1974(0)[a] 1975,1977(0)
Vallisneria americana	1926(H)				
Elodea canadensis			1958(H)		

[a] Aerial survey by Orth did not differentiate between *Zostera* and *Ruppia*

Tangier Island, Virginia

The MBHRL Survey team surveyed Tangier Sound once in the summer of 1972. *Ruppia* was found to be occasionally abundant but most often was present in trace amounts. *Zostera* appeared to be dominant and *P. pectinatus* was found to be abundant around Fishbone Island but was otherwise not present (see Table 63).

Table 63. Historic documentation of SAV by decade, Tangier Island, Virginia

Species	1970
Ruppia maritima	1972 (MB)
Potamogeton pectinatus	1972 (MB)
Zostera marina	1972 (MB)

Pocomoke Sound, Virginia

The Pocomoke Sound area was covered in 1972 by an MBHRL Survey crew. As would be expected of such a high saline area, Ruppia and Zostera were the only SAV species found. Zostera was the dominant species while Ruppia was found mainly in only trace amounts (see Table 64).

Table 64. Historic documentation of SAV by decades, Pocomoke Sound, Virginia

Species	1970
Ruppia maritima	1972 (MB)
Zostera marina	1972 (MB)

SUMMARY OF CONCLUSIONS

Submerged aquatic vegetation in the Chesapeake Bay has changed in species density, diversity and distribution over the past forty years (Renn 1937; Rawls 1964; Sculthorpe 1967; Lind and Cottam 1939; Elser, 1969; Holm et al. 1969; Bayley et al. in press). Some native species have disappeared, dominanance patterns have changed, drastic fluctuations in yearly SAV populations have been noted and seasonal patterns of growth, decline and reproduction have shifted.

Based on the lack of historic documented evidence for SAV in the Virginia portion of the Cheapeake Bay, conclusions based on data presented in this chapter pertain to Maryland Chesapeake Bay waters. Information for Virginia SAV is primarily limited to the work of Orth and Priest, working independently at the Virginia Institute of Marine Science.

During the first part of this century, the upper Potomac and Anacostia River estuaries were considered healthy ecosystems. But since that time, the flora and fauna of these upper tidal estuaries have diminished in diversity and density. In other parts of the Cheapeake Bay estuary, similar declines have been documented. Seven species of SAV were documented in the Sassafras River in the 1960s but only two have been found in the 1970s. The Susquehanna Flats that have served as a noted duck feeding ground no longer support the lush growths of SAV that have been documented over the past century. The Patuxent River supported at least eight species of SAV in the 1960s but only four species have been found in the 1970s. Around Curtis and Cove Points, four species have been documented since 1930 but MBHRL Survey teams have found nothing from 1971 to 1976.

Looking at the entire Maryland Bay, there is a general downward trend in vegetation occurrence and abundance, especially from 1971 through 1976. But, the losses appear to be random rather than predictable declines based on an epicenter concept or a large scale natural disaster. When data for decreasing species are combined for 1971 to 1976, there is no visible consistent trend of loss from any one point. Unlike some situations, such as the shellfish mortality in the Chester River in which increasing declines occurred down river in a gradient pattern, the SAV survey pattern does not indicate a strong point source mortality.

The basic questions which require consideration are whether Bay vegetation is (1) experiencing a normal population fluctuation, (2) responding to pollution or other man-oriented impacts or (3) both. The hypothesis of a normal cycle situation requires historical documentation of similar events in the past. But there is no documentation of such a significant portion of submerged flora having been so negatively impacted for such a long period of time. Only two species have been known for dramatic populations crashed in the past: Zostera and Myriophyllum. In both cases, other Bay species were able to replace them relatively quickly. This was a different system response than what has been seen in the 1970s when all species seem to have been negatively affected.

The variety and degree of human impacts, including a wide range of known or suspected pollutants, appear to be of greater relevance to recent declines in rooted aquatics than a naturally occurring cycle event. However, the hypothesis that the Bay is experiencing a cyclic phenomenon cannot be entirely ruled out. For example, the Bay has not appeared to be as resilient in the 1970s to excessive storms as was evinced during the 1930s. Hurricane Agnes seemed to have resulted in a greater negative impact of longer duration than comparable storms of the 1930s. Slower regrowth after Agnes may have resulted from negetive impacts incurred by the seagrass ecosystem due to man-oriented impacts and pollutants. Historic analysis of the upper Potomac River and Baltimore Harbor show once prolific vegetation areas now practically barren of floral species, semmingly due to excessive nutrient and pollution levels. Thus, if the Bay is experiencing a normal population fluctuation such a hypothesis would require consideration of human impacts on such a cycle.

Historic documentation of sufficient intensity and coverage necessary to support a cyclic phenomenon hypothesis is lacking. Therefore, those environmental parameters that have been impacted by man's activities in and around the Chesapeake Bay are discussed in the following chapter. Each parameter is discussed singly due to the mono-factorial nature of most experiments. In that the environment is more realistically a multifactorial system, it is likely that SAV declines have resulted from no single factor but from a multitude of factors or synergisms.

CHAPTER 4

ASSESSMENT OF ENVIRONMENTAL FACTORS

INTRODUCTION

There are various environmental and human related factors that affect the establishment, growth and reproduction of submerged aquatic vegetation in the Chesapeake Bay. Included among these are agrochemicals, salinity, turbidity, light, bottom substrate, nutrients, fauna, epiphytes, pH, temperature, heavy metals, petroleum products and water movement. Although there have been gradual changes in many of the environmental factors over the years, others, especially the human related factors, have changed more rapidly.

The dynamic nature of the Chesapeake Bay as a wind driven estuary naturally results in gradual changes in such parameters as salinity, bottom substrate and fauna. However, the changes that have resulted from increasing human impacts have been much more rapid. Increasing human population in the Bay area has affected land use patterns since 1900. Agricultural land use has decreased; fertilizer, lime and herbicide usage has increased; cropping practices have changed; wetlands have decreased in acreage by almost half. Rising population has also resulted in an increase in water, electric, sewage and industrial demands. In order to meet these demands, numerous water and sewage treatment plants, hydro-electric and industrial manufacturing facilities have been established in the Bay area. All of these facilities utilize the waters of the Bay and subsequently impact the estuarine system.

In order to assess the causes for the apparent declines in SAV, the various factors that have changed within the Bay area must be analyzed. This chapter presents these factors and attempts to selectively consider each one in order to determine the probable cause for Bay grass declines.

AGRO-CHEMICALS

Introduction

Land use patterns in Maryland and Virginia have changed drastically since the turn of the century. For Maryland alone, total land area of wetlands and agriculture has been halved since 1900 while forested land has increased slightly. Of major interest to this document is the decrease in agricultural land use (see Table 65) and associated changes in cropping practices. In order to

Table 65 . Total farmland in Maryland and Virginia, 1850-1974[a]

Year	Maryland[b] (hectares)	Virginia[c] (hectares)
1850	1,875,521	No information
1860	1,956,956	No information
1870	1,826,241	No information
1880	2,071,996	No information
1890	2,004,232	No information
1900	2,092,329	No information
1910	2,046,625	7,889,884
1920	1,925,562	7,511,682
1930	1,770,319	6,770,072
1940	1,698,861	6,655,254
1950	1,641,273	6,302,108
1959	1,398,954	5,312,012
1969	1,134,553	4,309,999
1974	1,079,611	4,492,170

[a] U.S. Department of Commerce 1974

[b] Approximate land area in Maryland = 2,574,799 hectares

[c] Approximate land area in Virginia= 10,428,180 hectares

compensate for the decline in land available for agricultural use, more efficient cropping methods have been developed. These changes have brought about increased usage of fertilizers, limes and new selective and contact herbicides.

No-till and minimum till cropping have become popular in the Bay area within the past several decades due to economic and conservation requirements. Though often referred to by other descriptive terms, these tillage techniques result in the least possible disturbance of the soil in order to effect seed placement, weed control and crop maturation (Shear 1965). Minimum soil displacement and maximum plant ground cover resulting from these techniques aid in the control of a varity of sound environmental practices: reduced wind and water erosion; increased soil organic content; reduced soil moisture evaporation; and increased infiltration of rainfall (Shear 1965; Bauemer and Bakermans 1972).

No-till and minimum till farming require the use of greater quantities of herbicides than conventional farming techniques. The presence of herbicides in agricultural runoff and the impact of herbicides to aquatic fauna and flora have recently become an issue for environmental concern, especially in an estuary such as the Cheapeake Bay where no-till has become popular. Increases in herbicide pound usage regardless of the source, have been implicated in the recent SAV declines of the 1970s. The role of no-till and minimum till farming as a source of water and soil borne herbicides is unknown at the present time. Despite the larger poundage rates that are necessitated with no-till farming compared to conventional farming techniques, no-till farming appears to result in reduced soil and water runoff. Since runoff is the major avenue for the introduction of herbicides into surrounding waters , no-till farming could possibly result in decreased amounts of herbicides reaching the aquatic environment. The extent to which erosion is decreased in relation to increases in herbicide poundage available to runoff has yet to be determined.

Fertilizers and Lime

The use of chemical fertilizers and lime have long been recognized for increasing crop yields. In Maryland, the tonnage of fertilizers used since 1935 has about doubled while lime usage has increased more than seven fold (see Table 66). Fertilizers have been used with direct aquatic application for the control of submerged plants. However, the possible impact to SAV of fertilizer and lime runoff as yet remains largely unknown.

Walker (1959) conducted pond studies on the distribution of water soluble fertilizer (N-P-K value of 16-20-0) applied at 22 to 38.5 kg/ha. Effective control of sago pondweed and mu kgrass resulted from a toxic reaction to the fertilizer as well as by shading from subsequent plankton blooms. Application of 110 kg/ha of 0-46-0 or 220 kg/ha of 8-24-8 sometimes stimulated filamentous algae growth. Of the three fertilizers studied, 0-46-0 and 8-24-8 did not control the submerged aquatic plants as effectively as 16-20-0.

Using plastic pools, Ryan et al. (1972) conducted a study to determine the effects of fertilization on the growth and mineral composition of three aquatic plant species; Eurasian watermilfoil, elodea and heartleafed pondweed. A

Table 66. Total fertilizer and lime used in Maryland, 1935-1976[a]

Year	Fertilizer (metric tons)	Lime (metric tons)
1935	149,644	54,215
1936	149,621	77,439
1937	169,109	106,180
1938	151,065	90,823
1939	149,410	89,502
1940	145,534	114,467
1941	156,088	117,375
1942	165,873	128,275
1943	180,503	176,279
1944	188,355	207,431
1945	198,041	208,368
1946	222,178	248,919
1947	232,351	266,788
1948	211,746	204,614
1949	222,610	199,626
1950	229,775	266,343
1951	249,571	281,085
1952	264,376	266,018
1953	269,309	251,512
1954	282,536	253,969
1955	281,829	267,571
1956	253,875	266,192
1957	269,211	Not available
1958	255,231	234,477
1959	278,970	260,191
1960	271,434	250,119
1961	269,971	253,985
1962	294,991	264,537
1963	323,593	274,419
1964	311,003	268,867
1965	327,534	328,010
1966	367,201	265,085
1967	345,425	236,230
1968	332,445	245,415
1969	334,018	254,085
1970	347,903	226,784
1971	335,819	180,617
1972	337,189	205,410
1973	400,825	251,786
1974	396,436	323,519
1975	340,810	293,511
1976	398,871	385,349

[a] Maryland Department of Agriculture 1963-1977

granular 8-8-2 fertilizer was used over two years and applied with three different treatments: a single application of fertilizer at a rate of 672 kg/ha; six weekly applications of the same fertilizer at 112 kg/ha; and an unfertilized control. The first year of the experiment the dry weight of milfoil was significantly greater in the control pools than in pools receiving the six weekly applications, due to competition from filamentous algae. Elodea and heartleaf pondweed were not adversely affected since they were able to grow through the mats of algae. During the second experimental year, both elodea milfoil controls had significantly greater yields than either fertilization treatments. Conversely, heartleaf pondweed had significantly greater yields in the treated pools.

Though available literature does not reveal information concerning the impact of fertilizers rich in runoff to SAV,Klausner et al. (1974) evaluated surface runoff losses of nitrogen and inorganic phosphorus from moderately to somewhat poorly drained soil as influenced by crop rotation, application rate and soil management practices. For the two year study, they found soluble nitrogen and inorganic phosphorus losses ranged from 0.39 to 29.23 and 0.04 to 0.49 kg/ha/year respectively. Annual surface runoff of NH_4-N was not significantly changed by crop, fertilizer application rate or soil management techniques,but was related to heavy fall application prior to a wet period. Inorganic phosphorus losses, in comparison, were related more to application rate and soil management than to time of application.

Increased levels of organic material in the water column,plus nitrogen and phosphorus, can give rise to eutrophic conditions where an aquatic ecosystem is enriched beyond its assimilation and flushing capabilities. Under such enriched conditions, minute algae plants or phytoplankton tend to thrive causing decreased light penetration. This shading effect from phytoplankton can be sufficient to actively decrease submerged aquatic vegetation. Such a situation has occurred in the Upper Potomac River where the water has been enriched far beyond the assimilation and flushing capabilities of the river. Agricultural fertilizers,whether from direct aquatic application or from indirect runoff sources,can provide the necessary enrichment for eutrophication.

Stolp and Penner (1973) conducted an experiment which suggested the possibility of increased phytotoxicity of herbicides when applied in conjunction with fertilizers. Though this study involved a terrestrial system, it is possible that similar results would occur in an aquatic system. Such a synergism should be considered in the investigation of herbicide and fertilizer runoff impacts to submerged aquatic vegetation.

Table 67 lists available data on fertilizer usage for counties in the Cheapeake Bay area. Of the eight Eastern Shore counties for which poundage rates were available, all but one showed increased usage from 1970 to 1976. Of the eight Western Shore counties, half showed increased fertilizer use.

The increased use of fertilizers around the Chesapeake Bay along with the potential impacts of fertilizers in an aquatic system necessitate the inclusion of fertilizers as a negative factor in an assessment of the effects of agrochemicals to the Bay.

Table 67. Summary of county fertilizer distribution data reported by registrants for 1970-1976

Year	Fertilizer (metric tons)							
	Eastern Shore counties							
	Cecil	Kent	Queen Annes	Talbot	Caroline	Dorchester	Wicomico	Somerset
1970	11,705	25,598	23,952	16,954	24,686	22,158	14,748	9,376
1971	10,955	24,419	26,006	15,781	24,723	20,574	16,048	8,807
1972	9,869	22,667	22,265	14,435	29,944	17,929	14,857	10,329
1973	11,324	25,889	33,954	17,139	30,512	24,229	18,207	10,937
1974	10,841	27,861	31,997	19,677	30,495	25,683	19,235	9,414
1975	9,531	28,542	28,449	16,322	26,907	25,369	19,640	8,118
1976	15,273	32,484	31,798	20,087	24,328	24,333	20,022	5,586

Year	Western Shore counties							
	Harford	Baltimore	Anne Arundel	Washington	Prince Georges	Charles	Calvert	St. Mary's
1970	15,327	14,367	6,493	12,034	10,832	7,365	4,852	6,619
1971	12,233	13,778	5,620	11,254	9,806	6,240	4,577	7,574
1972	13,302	17,623	5,736	13,074	14,154	8,689	5,401	9,798
1973	15,371	18,151	6,401	16,363	26,078	6,165	4,525	14,376
1974	16,244	14,883	7,005	16,400	25,611	8,026	4,763	8,424
1975	10,764	14,086	3,965	15,527	18,363	5,619	3,188	5,849
1976	17,338	13,652	4,950	22,894	25,309	5,713	4,139	10,181

Herbicides

The importance of herbicides to agriculture has given rise to a vast and current reservoir of avaiable information concerning mode of action, degradation pathways, breakdown products, persistence, toxicity, etc. under terrestrial conditions. Herbicides have also been used fairly extensively in research concerning direct application for the control of nuisance growths of submerged aquatic vegetation. What literature does not include, however, are the potential impacts of the indirect control of SAV from herbicide runoff. After land application, herbicides can potentially enter near-by waters through two mechanisms: leaching or dispersion into the dissolved portion of the water column; and adhering to soil runoff particles. Both surface and subsurface runoff can be affected.

Once on the soil surface the leaching potential of specific herbicides is dependent on the adsorptivity of the compound on varied soil colloids. Degradation pathways and by-product formation are important in determining the fate of herbicides in a complex ecosystem such as the Chesapeake Bay. To determine the long-range impacts of herbicides, their chemical persistence and toxicity to the flora and fauna must be understood.

The dislodgement of soil particles by the water-soil erosion process is termed "runoff". Concentrations of a given herbicide differ substantially in runoff based on differing soil types,land slope, application rate, weather and herbicide characteristics. For example, if soil moisture content is low, a high infiltration potential exists thus a brief rainfall of low intensity may not produce runoff but mobile pesticides may still move into the ground water system. Under saturated conditions, some herbicides may be leached downward in the soil profile, contributing to the subsurface runoff, dissolved in solution suspended in particulate matter; or adsorbed to the sediment (Pionke and Chesters 1973; Bailey et al. 1974b).

The most commonly used herbicides in the Chesapeake Bay area have been researched for information regarding degradation, volatility, fate in soil, toxicity and availability for runoff. Table 68 lists various general properties of the common herbicides and the following section discusses experimental work on terrestrial and aquatic ecosystems.

Cationic Herbidices - Diquat and Paraquat. Diquat and paraquat, cationic herbicides first described by Brian et al. (1958), are commercially available as dichloride and dibromide salts that readily dissociate in aqueous solutions. The herbicidal action of these compounds is dependent on the paraquat or diquat parent cation and results in the liberation of short-lived but active radicals, are responsible for herbicidal activity (Weed Science Society 1974).

Photochemical degradation. Photochemical breakdown of diquat and paraquat occurs upon exposure to ultra violet (UV) light (Slade 1965; Coates et al. 1966; Smith and Grove 1969; Funderburk et al. 1966). The two major degradation products of paraquat have been identified as 1-methyl-1-carboxypridinium ion and methylamine hydrochloride. The first degradation product displayed a low level of toxicity and degraded rapidly in soil and culture solutions (Calderbank 1968).

Table 68. Properties of commonly used herbicides

Herbicide	Chemical name	Water solubility 20 C - 25 C	Vapor pressure mm Hg 20 C (x10^{-6})	Molecular weight	Parachor
IONIC HERBICIDES					
I. Cationic					
Diquat (Orthodiquat)	6,7-dihydrodipyrido-(1,2-a:2,1-c)-pyrazidiinum dibromide	70%	very low	cation: 184.0 dibromide salt: 344	
Paraquat (Orthoparaquat)	1,1-dimethyl-4,4-bypyridinium-dichloride	70%	very low	cation: 186.2 dichloride salt: 257	
II. Basic					
Atrazine (Attrex)	2-chloro-4-ethylamino-6-isoprophlamino-s-triazine	pH 3: 31 ppm pH 7: 35 ppm	0.3	215.7	
Simazine (Princep)	2-chloro-4,6-bisiso-propylamino-s-triazine	pH3: 5.9 ppm pH 7: 5.0 ppm	0.0061	201.7	
III. Acidic					
2,4-D (Weedone 648)	2,4 dichlorophenoxy-acedic acid	650 ppm	at 160 C: 0.4	221.0	
Dicamba (Banvel D)	3,6-dichloro-o-anisic acid	4500 ppm	3570	221.0	
Dinoseb (Premerge)	2-sec-butyl-4,6-dinitrophenol	52 ppm	1.0	240.2	
NON-IONIC HERBICIDES					
IV. Substituted Anilines					
Trifluralin (Triflan)	ααα-trifluro-2,6-dinitro-N,N-dipropyl-p-toluidine	0.05 ppm	114	335.3	671
V. Phenylureas					
Linuron (Lorox)	3-(3,4-dichlorophenyl)-1-methoxy-1-methylurea	75 ppm	15	249.1	499
VI. Substituted Anilides					
Alachlor (Lasso)	2-chlor-2,6-diethyl-N-(methoxymethyl) acetanilide	148 ppm	22	269.8	626

Diquat in solution was found to degrade to 1,2,3,4-tetrahydro-1-oxo-pyrido-(1,2-α)-5-pyrazinium ion (Slade and Smith 1967). A second minor product resulted in the formation of pyridones that decomposed into volatile fragments (Ellis et al. 1956).

Little degradation occurred when paraquat in solution was exposed to sunlight because the absorption of UV light by aqueous paraquat occurs at 257 nm. The lower limit of the solar spectrum is 290 nm (Jordan et al. 1965 ; Slade 1966), thus aqueous paraquat stayed intact. Paraquat dichloride droplets photodecomposed when applied to maize, tomato and broad bean plants (Slade 1966). Contrasting paraquat, diquat readily decomposed in aqueous solution when exposed to UV light due to diquat's ultraviolet absorption band of 310 nm. Calderbank (1968) found the rate of photodecomposition of diquat sprayed on grass to be faster than for paraquat.

Haque et al. (1970) detected a shift of UV light absorption by aqueous paraquat to 278 nm when montmorillonite clay was added to solution. Funderburk et al.(1966) observed that photochemical degradation of paraquat adsorbed on soil and clay minterals (kaolinite and montmorillonite) occurred when exposed to UV light from a mercury lamp. In open soil situations paraquat and diquat can be quickly degraded but since there is little UV penetration in the tributaries of the Bay (Shea 1976), herbicide molecules are probably much more stable in the aquatic environment.

Volatilization. With exposure to UV light, both diquat and paraquat are degraded into volatile compounds when they are in dry form. Normally, however, the volatility of the bipyridylium compounds is considered low due to their low vapor pressure.

Fate in Soil. The efficacy of diquat and paraquat in no-tillage crop production is dependent on the rapid foliar necrosis of existing vegetation and subsequent loss of herbicidal activity. This rapid inactivation is common to the biphriylium herbicides. Brian et al. (1958) reported that diquat was water soluble (0.7 g/ml at 20 C) and that was rapidly adsorbed to soil particles. Detailed studies by Harris and Warren (1964) in which adsorption of chemicals from various solutions was measured, indicated that diquat was strongly adsorbed by organic soil, clay mineral bentonite and cation-exchange resins. This research further concluded that the removal of the diquat cation from solution in organic soil occurred by the ion exchange process. The herbicide cations displaced potassium ions from an organic soil saturated with potassium.

Calderbank and Slade (1975) cite the work of Malquori and Radaelli (1966) who compared the effectiveness of NH_4^+, K^+, Ca^{++}, Mg^{++} and Na^+ for releasing paraquat adsorbed to five different clays. Generally, K^+ and NH_4^+ were the most effective exchange ions; however, no paraquat was released when the concentration on clay was below a certain limit.

Soil adsorption of paraquat occurs in varying degrees of binding from unbound to tightly bound and often makes the herbicide unavailable for control of sensitive plants. The amount and strength of paraquat adsorption by soil depends on the amount and type of clay minerals present as shown above. For example, bipyridylium cations are tightly adsorbed in the latice of montmorillonite (Knight and Denny 1970) whereas herbicidal adsorption by kaolinite is more loose because adsorption occurs on the face of the clay particles (Coates et al. 1966).

At treatment levels much higher than normal application rates (0.25 to 1.0 µg/g soil), unbound paraquat has been found to remain on top of the soil and inhibit germination of such crops as corn and beans (Riley et al. 1976). The concentration of paraquat that inhibited germination varied according to soil type: 200 µg/g soil in sandy soil; 1200 µg/g in loamy sand; 2500 µg/g soil in loam, and 4000 µg/g soil in muck (Tucker et al. 1969). For most soil types, except sand, phytotoxic residues at higher doses are accounted for by assuming weak adsorption sites. These sites are available on soil surfaces after the stronger binding sites are occupied. Stevenson (1976) found that on organic soils and compost (14 percent organic matter) with low numbers of strong binding sites at the soil surface, a sufficiently strong salt solution may form during percolation to displace loosely held paraquat attached on weak adsorption sites, thus making paraquat available to sensitive plants.

Greater phytotoxicity ofparaquat on peat and muck does not accord with the behavior of most other herbicides in organic soils since organic matter tends to provide numerous adsorption sites. The strength and interaction between paraquat and organic matter is apparently sufficient to prevent leaching but may not prevent the stronger adsorption sites on a seed from adsorbing the paraquat bound to organic particles.

Paraquat is known to transfer not only from organic particles to seed particles but also to transfer from organic matter to sites on clays. Thus, paraquat is not leached in soils,yet is available to seeds sown on treated surfaces. Also, the phytotoxicity of diquat and paraquat in highly organic soils decreases as pH increases due to the anionic nature of the soil colloids with pH increase (Corbin et al. 1971). A comparison of diquat and paraquat adsorption suggests that their soil adsorption behavior is very similar. Although both herbicides can be adsorbed on clay, paraquat is preferentially absorbed and is more difficult to displace.

Hance (1967) studied the possibility of decomposition of paraquat in soil by nonbiological processes. Experiments using incubated soils were designed to inhibit biological degradation and promote chemical degradation. Under these conditions paraquat was stable. When paraquat was mixed in pots of soil, no loss of herbicide was reported after 14 to 16 months. Experiments with bibyridylium herbicides show that eventually strong binding sites on clay minerals are reached and degradation occurs only with difficulty.

Research by Fryer et al. (1975) indicated that paraquat displayed persistence in sandy loam soil. Paraquat applied over a period of seven years was totally extractable from the soil. Experiments by Weber and Cable (1968) showed that ^{14}C diquat retained on the interal surfaces of montmorillonite clay was not degraded by microorganisms. After one year the diquat was extracted in its original form. Diquat as well as paraquat adsorbed by montmorillonite could possibly persist indefinitely in its original molecular form.

Microbial decomposition. According to Funderburk (1969) the degradation pathway of paraquat by a bacterial isolate began with demethylation of the parent molecule followed by ring cleavage of one of the heterocyclic rings to form a final carboxylated n-methylpyridinium ion product. It is possible that other intermediate compounds form during degradation but they have not been recorded.

Baldwin et al. (1966 cited in Funderburk 1969) studied numerous organisms in synthetic media for the ability to use paraquat as a sole carbon or nitrogen source. *Corynibacterium fascians* and *Clostridium pasteurianum* were found to decompose the paraquat molecule at variable rates. A soil yeast, identified as *Lipomyces starkeyi*, decomposed paraquat and demonstrated a preference for paraquat over nitrate nitrogen. When montmorillonite clay was added, degradation by the yeast organism was observed to cease (Weber and Cable 1968). Utilizing labeled paraquat, Burns and Audus (1970) determined that cultures containing high organic soil demonstrated significant paraquat decomposition by the yeast compared to soils with a lower organic content.

Aquatic Weed Control. Diquat and paraquat were observed by Blackburn and and Weldon (1964) to be two of the most promising herbicides for the control of *Najas guadalupensis* in irrigation channels in Florida. Paraquat was applied at 5 and 10 ppm while diquat was applied at 1.5 and 2.5 ppm. After eight weeks both herbicides effected 100 percent weed reduction. Approximately 20 weeks following initial treatment, there was still 80 to 85 percent growth inhibition.

Yeo (1966) found concentrations of 250 ppb diquat controlled sago pondweed, American elodea and common naiad for eight weeks or longer. Curlyleaf pondweed was controlled for four to six weeks with increasing concentrations of diquat up to 1,000 ppb. Coontail and American milfoil were eliminated at 500 ppb but *Chara* appeared to be tolerant of diquat up to concentrations of 1,000 ppb. Coontail and American milfoil were eliminated at 500 ppb but *Chara* appeared to be tolerant of diquat up to concentrations of 1,000 ppb. Elodea, coontail, and duckweed were controlled with 500 ppb diquat in growth pools. Paraquat was found to control sago pondweed and American elodea at a concentration of 250 ppb when applied in reservoirs. In growth pools 1,000 ppb paraquat controlled coontail, American elodea and sago pondweed.

Once incorporated in ponds, diquat has been found to dissipate rapidly (Coates et al. 1964; Frank and Comes 1967; Grzenda et al. 1965; Yeo 1966). It becomes readily bound to sediments, suspended particulate matter and to the surfaces of aquatic plants due to its cationic binding properties. Another manner in which diquat is removed from the water column is through uptake by aquatic plants and algae (Newman and Way 1966; Davies and Seaman 1964).

A study by Funderburk and Lawrence (1963) of the absorption and translocation of radioactive herbicides in water star grass (*Heteranthera dubia*) indicated that diquat and paraquat were taken up by both roots and shoots. Radioassays detected slight movement of root-applied herbicides to the upper portion of the stem. There was no movement in the downward direction. Other aquatic weed species, such as *Elodea canadensis* and *Potamogeton pectinatus*, have been shown to absorb root-or foliar-applied diquat (Funderburk and Lawrence 1963; Davies and Seaman 1964.

A study by Simsiman and Chesters (1975) indicated that with 1.5 μg/ml diquat applied to weeds of an infested impoundment, after 22 days, 19 percent of the chemical remained in the sediment. The herbicides associated with the decayed plants tended to remain in the organic layer if no sediment incorporation occurred. Following rapid weed kill, profuse proliferation of micoorganisms was found which may have helped in the degradation of diquat. Surges in bacterial numbers have also been observed in water immediately after paraquat treatment in weed infested reservoirs and lakes (Way et al. 1971; Fry et al. 1973). Since diquat and paraquat are weakly bound to decomposed weeds, these chemicals are more susceptible to microbial degradation than if they were adsorbed on clay particles. Slow microbial degradation of sediment adsorbed diquat was demonstrated by Simsiman and Chesters (1975) After 0.3 kg/ha diquat application in a pool, a maximum of 1.7 μg/g was found in the sediment after four years.

Frank and Comes (1967) studied herbicide residues of diquat and paraquat in pond water and soil. High concentrations of paraquat persisted for several days in the water column but by the 12th day, adsorption by soil appeared to be complete. After 85 days, concentrations of paraquat were found in the soil fraction. Diquat acted similarly but still persisted in high concentrations in the soil after the 160-day experiment.

Basic Herbicides - Atrazine and Simazine. Atrazine and simazine are members of the herbicidal group called the s-triazines. This group characteristically consists of basic compounds that become associated with hydrogen-forming protonated complexes in solution (Allcook 1967). The quantity of complexes formed is governed by equilibrium forces.

The herbicidal action of s-triazine compounds was first discovered in 1952 by a group of researchers from J. R. Geigy, Ltd. in Switzerland (Gysin and Knusli 1954, cited in Esser et al. 1975). In 1956, a product specified as simazine was described and found to be highly selective with long-lasting residual durability. Atrazine, today's primary s-triazine for weed control in corn, was developed and released for experimentation in 1957 and became commercially available in 1958.

The mode of action of these s-triazine compounds is to inhibit the photosynthetic Hill reaction, thus irregulating the photosynthetic mechanism (Gysin and Knusli 1960). Atrazine and simazine are widely used for control of broadleaf and grassy weeds in corn and other crops (Weed Science Society 1974). These herbicides became popular due to their outstanding tolerance by corn and effective spectrum of weed control.

Roeth and Lavy (1971), cited in Esser et al. (1975) found that extracts from corn mixed with atrazine or simazine formed some of the same hydroxy derivatives when incubated. The inability of most plants to hydrolyze chloro-s-triazine compounds causes these chemicals to have a broad spectrum of weed control. Discoveries revealed that the compound in corn which is capable of inactivating the simazine or atrazine molecule is 2,4-dihydroxy-methoxy-1,4 (2H)-benzoxazin-3 (4H)-one (DIMBOA) (Palmer and Grogan 1968). Corn is capable of forming hydroxyatrazine as the predominant degradation product.

Photochemical Degradation. Atrazine, when subjected to UV light, undergoes photochemical decomposition which brings about a color change of white to tan. Far-UV irradiation has been demonstrated to cause the greatest chemical and physical change. (Jordan et al. 1964; Jordan et al. 1965). Different reaction products have been isolated depending on the environmental conditions of the experiment. In aqueous solution 2-chloro-s-triazines degraded to 2-hydroxy analogs; however, in alcholic solution, 2-alkoxy derivatives were formed (Nikles and Ekner 1963).

Loss of atrazine and simazine is rapid following light exposure; this reaction decreases with time. This decrease is caused by a masking effect which occurs with the build-up of decomposition products that partially protect intact herbicide molecules from photodegradation. Comes and Timmons (1965) found that atrazine and simazine directly sprayed on soil were detoxified by exposure to sunlight. Exposure of atrazine caused a 47 percent atrazine loss in 25 days in the spring and 65 to 80 percent loss in summer. Simazine loss due to irradiation in spring was 25 percent after 25 days. However, volatilization may have occurred simultaneously decreasing the actual amount of chemical decomposition thought to be caused by photolysis. Herbicide degradation in control soils where atrazine was incorporated into the soil and thus not exposed to light was negligible, indicating that atrazine had greater persistence when buried beneath the surface.

Volatilization. As mentioned above, herbicidal s-triazines have been shown experimentally to be subject to volatilization. Volatility loss is dependent on soil type, temperature, moisture content and the physical and chemical nature of the individual s-triazine compound. Weed Science Society (1974) lists atrazine as having a vapor pressure of 0.3×10^{-6} mm Hg at 20 C indicating slight volatility. Simazine's vapor pressure is 0.0061×10^{-6} mm Hg at 20 C which suggests a relatively low rate of volatility.

Kearney et al. (1964) applied several s-triazines to Tifton loamy sand at a rate of 3.3 kg/ha to record losses resulting from volatilization. Results indicated that simazine was a fairly stable compound with slight volatilization occcurring at 35 C. Davis et al. (1959) reported that 50 percent of simazine originally applied to a metal surface volatilized at 71 to 73.5 C with less volatilization occurring at lower temperatures.

Within the s-triazine group, atrazine is one of the most volatile s-triazine compounds. Kearney et al.(1964) found losses of atrazine from all soils studied with a volatility most rapid from light textured soils. Increasing temperatures in 10 C increments tended to double the loss of atrazine. After 72 hours at 45 C, more than half of the atrazine was lost from Bosket loam (51 percent sand, 42 percent silt, 7 percent clay, 0.6 percent organic matter) and Cecil sandy loam (66 percent sand, 21 percent silt, 14 percent clay, 0.8 percent organic matter).

Kearney et al. (1964) also found that moisture content of soil affected the rate and quantity of herbicide volatilization though simazine loss from moist soil was small compared to atrazine. On dry soils, simazine loss increased indicating that it is more volatile under dry conditions. Differences in volatility due to moisture content was attributed to the water solubility of the various s-triazines, their penetration depth in soil and basic differences of herbicide adsorption to binding sites in the soil.

Fate in Soil. Residual chloro-s-triazines persist in soil. The least amount of leaching was found by Nearpass (1965) to occur in organic or heavy textured soils versus light textured soils. Nearpass concluded that organic matter content of soil was correlated with adsorption of simazine. A decrease in the mobility of s-triazine in soil reflected an increase in their relative adsorption.

Simazine was found to have greater mobility in sandy soil than in clay soil (Stroube and Bondarenko 1960). Accordingly, atrazine was mobile in a fine sandy loam with less leaching from Drummer clay loam (25 percent sand, 45 percent silt and 30 percent clay). Clay has been found to nearly cease the mobility of simazine in 24 percent clay content soils (Gast 1959).

S-triazine movement in four Maryland soils was observed by Harris (1966) utilizing soil columns subjected to an upward free-flowing water supply. Atrazine moved to the upper segments of the experimental columns indicating a leaching potential. Although often thought of as degrading quickly, atrazine residues have been found in the subsoil as well as top soil for extended periods after planting. Burnside et al. (1963) found 0.8 ppm atrazine at 45 to 60 cm depth of silty clay loam soil 16 months after a 2.75 kg/ha treatment.

Experiments by McGlamery and Slife (1966) indicated that pH affects adsorption to a greater extent than temperature on Drummer clay loam soil. Atrazine adsorption increased markedly as pH decreased below pH 6; desorption increased as pH and temperature increased.

Weber (1970) and Weber et al. (1969) found adsorption on hydrogen saturated montmorillonite clay occurred as the basic triazine molecule complexed with hydrogen ions on the adsorbant surface. Triazine molecules desorbed from clay with the introduction of acids, bases or salts to the bound system.

Normally, clay and organic matter function as a unit to adsorb the s-triazines. Weber (1970) found that the interaction of organic matter with clay provided an inorganic surface for adsorption. In the final analysis, however, the quantity of organic matter surrounding the clay particles regulated the adsorptive capacity of the soil. It was determined that different types of organic matter had varying numbers of reactive groups available for hydrogen bonding. The prevalence of these reactive groups was concluded to be responsible for differences in relative adsorption values in soils of high organic matter content.

Microbial Degradation. The major mechanism for microbial degradation of the chloro-s-triazines is dealkylation. Kaufman and Kearney (1970) also found microbial degradation to occur through hydroxylation and ring cleavage with subsequent CO_2 release. The soil organisms effective in degrading atrazine and simazine included various genera of bacteria and fungi.

Roeth et al. (1969) studied the effects of temperature, moisture and microorganisms on the degradation of atrazine. Degradation occurred two to three times more quickly in top soils compared to subsoils and increased by a factor of two to three with 10 degree temperature increments from 15 to 35 C. Increasing soil moisture content further augmented ring cleavage of ^{14}C labeled atrazine.

Goswami and Green (1971) studied the degradation of atrazine and its hydrolysis product, hydroxyatrazine, under laboratory conditions of limited aeration similar to those found in marine and estuarine sediments. ^{14}C-ring labeled atrazine was added to 5.0 g soil to give 10 μg atrazine/g Kapaa soil and 20 μg atrazine/g Molokai soil. Degradation was measured in submerged Kapaa soil (10.5 percent organic matter, 31.8 X 10^6 bacteria/g soil, pH 4.7) and submerged Molokai soil (3.4 percent organic matter, 1.4 X 10^6 bacteria/g soil, pH 6.4) by measuring $^{14}CO_2$ evolution. In 30 days, 0.02 percent of the added atrazine evolved from the Molokai soil and 0.59 percent from the Kapaa soil. The higher percentage of organic matter and microbiological population present in the Kapaa soil was hypothesized to have resulted in enhanced degradation of hydroxyatrazine to CO_2.

Runoff Losses. In a field experiment designed to maximize runoff losses, Hall et al. (1972) applied from 0 to 9.0 kg/ha atrazine as a pre-emergent to corn plots of Hagerstown silty clay loam on a 14 percent slope. With increasing rate of herbicide application water and soil runoff generally increased as did levels of atrazine in runoff water. Twenty-three days after herbicide application, the first rainfall event yielded atrazine concentrations from 0.39 to 4.68 ppm in runoff water. A subsequent rainfall nine days later yielded concentrations of atrazine in runoff water from four to five times less. Generally, total atrazine loss was proportional to the water loss during the growing season. Despite occasional higher concentrations in the sediment, Hall et al. (1972) recovered greater quantities of atrazine from the water fraction compared to the sediment portion. Atrazine lost in the water fraction ranged from 5.0 to 61.0 g/ha with increasing herbicide application rate. The amounts lost from soil sediment on the same date ranged from 1.2 to 3.0 g/ha. Greatest percent loss of total applied atrazine occurred in runoff water with a treatment rate of 1.1 kg/ha with 3.67 percent loss and the lowest was at 0.6 kg/ha with a loss of 1.7 percent. Losses in eroded sediment increased from 0.03 to 0.28 percent as application rates increased from 0.6 to 9.0 kg/ha. Hall et al. (1972) concluded that during the early growing season, the amount and frequency of rainfall were particularly important in relation to runoff losses. Mid and late-season water losses appeared to have been influenced by crop density and evapotranspiration. Concerning the contamination of waters adjacent to areas treated with atrazine, Hall et al. concluded that impacts would probably be minor providing atrazine was applied at recommended levels and good soil and crop practices were maintained.

Bailey et al. (1974a) studied soil type and application rates for atrazine and dichlobenil (DCBN) in relation to runoff from a simulated 100-year frequency storm event. The two herbicides were applied simultaneously to four Coastal Plain soil types according to Table 69.

Sediment and water fractions were measured at five-minute intervals during the two-hour simulation. Herbicide concentrations in both fractions were highest during the first 40 to 50 minutes of the storm, then decreased unevenly over the next hour. In attempting to explain this uneven decrease in concentrations of both herbicides, Bailey et al. (1974a) concluded that initial high runoff concentrations resulted in the removal of the "zone of erodibility." Once leaching action depleted this layer of herbicide concentration and soil removed by storm-related erosion, losses of atrazine and DCBN decreased. Herbicide runoff during this lower concentration period resulted from the diffusion of herbicides up to the laterally moving layer from a lower pesticide-rich zone. A second herbicide runoff concentration increase occurred when this pesticide-free zone was removed exposing the pesticide -rich zone to erosion. Variations in this "plateau" effect resulted from varying herbicide solubility, soil permeability and infiltration characteristics.

In comparing runoff by soil type, Bailey et al. (1974a) found that herbicide loss was less from the Dothan plot. Herbicide loss appeared to be related to loss of clay-sized soil particles and water. Average losses of atrazine and DCBN showed a loss of 10.7 percent total applied atrazine compared to a loss of 6.6 percent total applied DCBN. Individual adsorption characteristics of the two herbicides were not determined to be dominant in relation to runoff.

Recent runoff studies from an agricultural watershed within the Chesapeake Bay area involved surveillance of a cornfield in the Rhode River watershed (Wu et al. 1977). The plowed zone of this silty loam soil type was 3 to 0.15 percent organic matter. For this soil type runoff loss of atrazine was calculated to be 1.2 percent.

Recent field sampling by Correll et al. (1977) in several Chesapeake Bay estuaries (discussed in detail later) indicated that atrazine had been found at concentrations up to 8 ppm in sediments. Bioassays by Correll et al. showed that while 1 ppm atrazine in sediments appeared to have no negative impacts to Zannichellia palustris, at a 10 ppm level in sediments, atrazine effected decreases in net O_2 production and gross photosynthesis.

Table 69. Atrazine and DCBN applications to four Coastal Plain soil types[a]

Soil Type	Slope (percent)	Atrazine kg/ha	DCBN kg/ha
Dothan sandy loam	2.2	1.68	6.72
Red Bay sandy loam	2.5	3.36	6.72
Malbis (Bowie) sandy clay loam	3.6	1.68	6.72
Malbis (Bowie) sandy clay loam	5.7	3.36	6.72

[a] Bailey, et al. 1974a

Aquatic Weed Control. Atrazine and simazine first tested for aquatic weed control in Missouri farm ponds by Walker (1964). In one pond, atrazine was applied at 0.2 to 6.0 ppm to 11 submersed vascular plant species and four species of filamentous algae. Seasonal growth inhibition was generally exhibited at low concentrations (0.5 ppm). Complete erradication for more than a year was achieved at higher application rates (1.0 ppm). Specific data indicated that an application rate of 1.0 ppm atrazine controlled Najas flexilis in more than 50 percent of the open treatment plots. Out of 414 tests, approximately 66 percent control was achieved in whole ponds and plastic enclosures, and 22 percent control was found in open plots. The granular formulation of simazine was determined to provide control of rooted aquatics, while the wettable powder was a more satisfactory inhibitor of algae.

Norton and Ellis (1976) demonstrated that simazine was highly effective and selective at controlling blue-green algae. Rates required for algal control were 0.25 ppm in thermally stratified lakes. In non-stratified lakes, poor planktonic algal control resulted since simazine was not contained above the thermocline as in stratified lakes. Since simazine was incorporated in the hypolimnion zone, double the concentration (0.5 ppm) was needed for algal control. Norton and Ellis found with both 0.5 and 0.25 ppm treatment levels that Chara spp., Myriophyllum spp., Vallisneria americana and Potamogeton spp. present in lakes were not eliminated but exhibited growth suppression.

Since simazine affects the Hill reaction in photosynthesis, it results in a reduction in the amount of dissolved oxygen in the aquatic environment. Nutrient cultures of Elodea canadensis treated with 0.12 to 1.0 ppm simazine showed signs of inhibited O_2 evolution within 24 hours (Sutton et al. 1969). At the 1.0 ppm treatment level, normal evolution of 12.0 to 14.0 ppm dissolved O_2 was reduced to 8.0 ppm O_2. Even at a low treatment of 0.12 ppm simazine caused a significant reduction of dissolved oxygen. Sutton et al. (1969) concluded that decreased photosynthesis of aquatic plants after chemical treatment and the decrease of oxygen resulting from decaying vegetation suggested a deterimental impact to aquatic fauna and flora.

Funderburk and Lawrence (1963) conducted experiments to determine the mode of uptake of s-triazines. Autoradiographs of a labeled simazine-treated submersed weed, waterstargrass (Zosterella dubia), indicated that both shoots and roots were able to take up simazine. Movement of the carbon-labeled herbicide from roots to shoots and vice versa was observed. Significant downward translocation of simazine has also been observed in terrestrial plants (Funderburk and Lawrence 1963).

The characteristic herbicidal effect of both the atrazine and simazine treatments is a chlorotic appearance and progressive decomposition of affected SAV plant parts (Walker 1964).

Acidic Herbicides--2, 4-D, Dicamba and Dinoseb. Acid herbicides ionize in aqueous solution to yield anionic groups and include the chloro-phenoxyacids (2,4-D), substituted benzoic acids (dicamba) and the weakly acidic phenols (dinoseb). The chloro-phenoxyacids were introduced as selective weed-killers at the end of World War II and are unique due to their high activity against many broad leaved species but not against graminaceous species (Peterson 1967). Acidic 2,4-D is readily absorbed by both leaves and roots and is concentrated at the root and shoot meristems. Acidic 2,4-D causes abnormalities in growth, respiration, food storage and cell division (Weed Science Society 1974). Of the benzoic acids, dicamba was introduced in the early 1960s as a selective herbicide for pre-emergence and post-emergence growth regulation of annual broadleaf and grassy weeds in cereals (Velsicol Chemical Corp. 1967). The acidic phenols are among the oldest known organic pesticides (Kirby 1966). Dinoseb controls seedling weeds and grasses and established perennial weeds (Weed Science Society 1974). Dinoseb causes cell necrosis through direct leaf contact.

Photochemical Degradation. These three acidic compounds degrade photochemically. Irradiated solutions of isopropyl and butyl esters of 2,4-D in distilled water were degraded to 2,4-dichloro-phenol at a rapid rate (Ellis et al. 1941). Bell (1956) found that the concentration of 2,4-D decreased continually after exposure to UV light. Photodecomposition of 2,4-D compounds proceeded at relatively slow rates at pH 4.0 and more rapidly at pH 7.0 with a maximum rate reached at pH 9.0. Dinitrophenols, including dinoseb, also appear to be more stable in acidic solutions. However, they are susceptible to decomposition by UV radition in alkaline solution (Molnar 1935, cited in Kaufman 1976).

Volatilization. Dicamba is considered stable and non-volatile. The volatility of 2,4-D depends on the formulation used. The least volatile are the oil soluble amines. Inorganic salt and acid formulations are less volatile than the esters which can vary from low to high volatility (Weed Science Society 1974).

Dinoseb is reported to volatilize from soil surfaces (Davis 1956). This latter observation is consistent with dinoseb's relatively high vapor pressure of 1.0×10^{-6} mm Hg (at room temperature). Dinoseb also exhibits a higher rate of volatilization in moist soils with increasing temperature. Plant kills by dinoseb vapors have been reported in literature (Hollingsworth and Ennis 1953, cited in Kaufman 1976; Davis and Selman 1954; Swanson et al. 1953). Hollingsworth and Ennis (1953) found that a two to three degree temperature change between 28.8 and 35.5 caused an 18 to 69 percent increase in plant mortality due to increasing vapor activity.

Fate in Soil. Weber (1972) found the mobility of acid herbicides to be highest in coarse textured sandy soils compared to fine clay or organic soils. Organic matter content appears to be one of the most important soil factors related to acidic herbicide adsorption. Also, this adsorption is reversible. Weber et al. (1965) found that three extractions of 1M NaCl were required to release 2,4-D adsorbed to strongly basic anion exchange resins. Negative adsorption or repulsion of acid anions by clay colloids occurred at neutral or basic pH levels. In strongly acid systems, positive adsorption of molecular species occurred. In limed or naturally alkaline soils, 2,4-D toxicity was retained for a longer period of time than in unlimed soils (Kries 1947). 2,4-D adsorption by clay has been demonstrated by experiments that withdrew the water from the complex, leaving the chemicals to adhere to silicate surfaces (Harter and Ahlrichs 1969). The bonding, however, was weak and exposure to low levels of moisture quickly moved the chemical into the soil column.

Dicamba exhibited intermediate persistence in many soils when compared to other acidic herbicides. However on a variety of soil types, dicamba was found to be highly mobile (Donaldson and Foy 1965) depending on water flux and capillary water movement (Harris and Warren 1964). Adsorption by most clays was limited but acidic kaolinite clay and muck soils indicated extensive adsorption of dicamba (Donaldson and Foy 1965). This is especially important in the Chesapeake Bay area where kaolinitic clay is prevalent.

Soil pH levels are directly related to adsorption of dicamba, with strongest binding capacity at lower pH levels (4.0 to 6.0) and minimal at pH levels higher than 6.0 (Corbin et al. 1971). Water soluble salts of dinoseb have been reported to leach in soil where the oil-soluble or water miscible forms displayed less movement. The leachability of dinoseb in soil was affected by soil texture, moisture content and herbicide formulation (Weed Science Society 1974).

Dinoseb is similar to dicamba in that adsorption in soil is pH dependent . Weber et al. (1965) found a lack of dinoseb adsorption on bentonite at pH 8.4. However, the acid herbicides including 2,4-D, dicamba and dinoseb were found generally adsorbed on an acidic muck at relatively low amounts compared to other herbicides (Weber et al. 1965). The capacity of these herbicides to be adsorbed was inversely related to water solubilities.

Microbial decomposition. Research (Mitchell and Marth 1946; Brown and Mitchell 1948) has revealed that 2,4-D is rapidly detoxified in soil as compared to other phenoxyacetic acids. High rates of 2,4-D microbial decomposition were enhanced by warm soil conditions with high organic matter content. Inhibition of 2,4-D degradation in autoclaved soil indicated that the decomposition pathway was microbiologically induced.

The microbial pathways of 2,4-D decomposition have been investigated by Audus (1964). Complete or nearly complete degradation of several phenoxyacetic acids was accompanied by loss of aromatic structure and release of chlorine or chloride ions. There appear to be two main pathways for the degradation of phenoxyacetic acids: degradation via a hydrophenoxyacetic acid intermediate; and degradation via the corresponding phenol (Loos 1969). Evans et al. (1971) suggested that degradation by the first pathway resulted in production of a phenolic acid metabolite (6-OH-2,4-D). A fairly low pH optimum of 5.3 for 2,4-D was described. Fungi are thought to be the agents for degradation in the highly acidic soils made more basic with lime (Corbin and Upchurch 1967).

Elad et al. (1965) established that the degradation pathway of dicamba, metabolized by microorganisms, involved 5-hydroxy-2-methyl-3,5-dichloro benzoic acid as a major conjugated metabolite. One minor degradation product was 3,6-dichloro salicyclic acid. The reactions involved in photodegradation of dinoseb included side chain hydrozylation.

Runoff from Coastal Plain Soils. 2,4-D has been detected in streams by a Federal monitoring program (Schultz et al. 1973). Research has indicated that esters of 2,4-D are lost more readily in runoff than the anion salts (Evans and Driseja 1973).

A three-year cornfield study in the Georgia Coastal Plain was conducted by White et al. (1976) to determine soil retention of 2,4-D with simulated rainfall using a 0.34 ha study area of Cowarts loamy sand with 0.5 percent organic matter content. A zone of low hydraulic conductivity existed at a depth of 92 to 214 cm, which enabled researchers to evaluate subsurface as well as surface water quality outputs. When rains were applied, 1, 8 and 35 days after herbicide application (0.56 kg/ha), total 2,4-D concentrations for soil (sediment) and water fraction combined were 25.2, 5.8 and 0.7 μg/l respectively. The maximum concentration of 2,4-D in surface runoff was 8.1 μg/l compared to 1.2 μg/l for subsurface runoff. In comparing maximum surface runoff concentration with other similar work (Evans and Driseja 1973; Sheets and Lutz 1969) White et al. found their own results to be lower. It was concluded that under the conditions of this watershed, subsurface flow was three times larger than surface flow. However, in the surface 0.5 cm of soil, 2,4-D concentration decreased 95 percent in the first 7 days and reached 0.01 ppm after 34 days. 2,4-D was not found to accumulate in soils applied at a 90 cm depth. The potential for 2,4-D runoff from rainfall events occurring soon after herbicide application may have important implications for subtributaries of the Chesapeake Bay where 2,4-D is commonly used.

Aquatic Weed Control. Applications of 2,4-D were used extensively in the Chesapeake Bay in the 1960s to control milfoil infestations. Effective control of milfoil was attained between the latter part of May and the first half of June using 22 to 33 kg a.e./ha (Rawls 1965, 1971*b*, 1975). The most effective method of control using 2,4-D (BEE) involved first mowing the underwater plants followed by herbicide application at low tide, just before low water slack (Steenis and Stotts 1965). Large scale operations using helicopters have been used to apply 2,4-D (BEE) in locations where there was no commercial fishing (Rawls and McKee 1964). These treatments were tested and found to be non-toxic to sport or commercial fish. However, low levels of 2,4-D were known to temporarily concentrate in oysters and clams, and so extensive use of this treatment was curtailed (Beaven et al. 1962; Rawls and Beaven 1963; Steenis and King 1964).

Tests by Aldrich and Otto (1959) using *Potamogeton pectinatus* indicated that both leaf-fed and root-fed plants accumulate 2,4-D-1-C^{14}. Apparently, translocation proceeded more rapidly downward than upward. In labeled 2,4-D butxyethyl ester treated waterstar grass, again root and shoot uptake was observed with downward translocation predominating (Funderburk and Lawrence 1963).

Non-ionic Herbicides (Substituted Dinitroaniline)-Trifluralin. The dinitroanilines are highly selective herbicides introduced in the 1960s (Brooks et al. 1960). One of the most important members of this class of herbicides is trifluralin which was first registered for use on food crops in 1964 (Probst et al. 1976). This 2,6-dinitroaniline exhibits preemergence herbicidal activity selectively toward grasses rather than broadleaf weeds and has become popular due to its effective control of Johnsongrass (Weed Science Society 1974; Probst et al. 1976). It controls seed germination and associated physiological growth processes (Weed Science Society 1974). A wide variety of crops are resistent to trifluralin making it useful in crop rotation. Trifluralin is usually incorporated into the soil rather than applied to foliage. There is no significant translocation or adsorption of trifluralin by soil treated crops (Weed Science Society 1974).

Photodecomposition. Dinitroanilines are very unstable when exposed to ultraviolet light. The photodecomposition pathways of trifluralin involve dealkylation, cyclization, reduction and oxidation. Numerous degradation products are formed when trifluralin is exposed to sunlight wavelengths. Leitis and Crosby (1974) found that under acid conditions, dealkylation produced a primary degradation product of 2-amino-6-nitro α,α,α,-trifloro-p-toluidine. Under alkaline conditions the major photolysis product was 2-ethyl-7-nitro trifloro-methylbenzimidazole. Application of trifluralin under field conditions, especially where the chemical might be incorporated in irrigation water, revealed rapid photodecomposition to numerous other substances.

Extensive field trials have shown that when trifluralin was incorporated into the soil, a marked enhancement (3 or 4 times) of herbicidal activity took place (Ketchersid et al. 1969). Apparently, trifluralin was either rapidly inactivated by a photochemical process or was volatilized from the soil surface. Further, once trifluralin volatilized, it could be photolytically degraded in the atmosphere.

Volatilization. In a laboratory study relating vapor losses to soil moisture, Parochetti et al. (1976) found trifluralin very volatile compared to ten other dinitroaniline herbicides. Trifluralin was tightly adsorbed to soil particles. At 50 C, vapor losses decreased from approximately 23 percent at 7 percent soil moisture to 17.5 percent at 15.3 percent soil moisture. Increasing soil temperature was found to enhance volatility. This corroborated earlier work by Spencer and Cliath (1974) who found trifluralin vapor pressure to increase approximately five times for each 10 degree temperature increase between 20 C and 40 C.

In a similar laboratory experiment, Harvey (1974) found trifluralin to be relatively volatile compared to other dinitroaniline herbicides. Neither soil texture or type greatly influenced the phytotoxicity of trifluralin to the plants studied; however, increasing soil moisture decreased volatility. Based on *in vitro* experiments with foxtail millet , Harvey found the absorption of trifluralin vapor to cause substantial root growth reduction. He concluded that vapor absorption may have been a more effective mode of plant entry than herbicide absorption from soil solutions.

In comparing trifluralin volatilization from aerobic and anaerobic soils in a moist or flooded state, Paar and Smith (1973, cited in Probst et al. 1975) found flooding to greatly decrease volatility. Volatilization from moist aerobic soil after 20 days reached a cumulative total of 2.5 percent compared to less than 1.0 percent from moist anaerobic soil.

Bardsley et al. (1968) found trifluralin to be highly volatile in an aqueous media. They determined that water vapor promoted volatilization not only from a water media but from the soil. These results were corroborated by Weber (1972) who concluded that increasing volatilization of trifluralin occurred with increasing soil moisture possibly due to water vaporization.

Fate in Soil. The dinitroanilines are among the least mobile herbicides (Harris 1967) and are readily adsorbed by organic matter (Parka and Tepe 1969). Adsorption by soil organic matter has been related to the parachor of these compounds. Generally, larger molecules are adsorbed more readily than smaller ones. Trifluralin has a parachor of 671 which is the largest of all non-ionic herbicides. Accordingly, it is not as readily available for weed control as the more loosely soil bound herbicides (Lambert et al. 1965 cited in Weber 1972). Parka and Tepe (1969) studied residual trifluralin from 107 locations in the United States where herbicide applications had been made for one to four years. It was concluded that trifluralin did not accumulate in the soil even after repeated applications.

Miller et al. (1975) investigated the soil persistence of trifluralin applied to a fine sandy loam soil over a period of five years. Residues were mainly in the upper 30 cm of soil with 80 percent concentrated in the top 15 cm of soil. Fifteen months after final application, trifluralin residues were greatly reduced in the top 15 cm and almost nonexistent after 30 months. Degradation of trifluralin in soil 15 to 30 cm deep progressed more slowly. However, herbicide levels at this depth never exceeded trace amounts. The relative adsorption of trifluralin by soil organic matter was similar to the cationic herbicides. Experiments using montmorillonite clay and organic muck soils indicated that phytotoxicity was decreased by increasing soil organic matter, but not by increasing clay (Weber 1972).

The degradation of trifluralin in soil is affected by anaerobic and aerobic conditions as well as soil type, moisture and temperature. In comparing aerobic and anaerobic degradation, Probst et al. (1975) discussed an experiment utilizing 0, 50, 100 and 200 percent field capacity moisture. Trifluralin degradation was greatest under the anaerobic conditions of 200 percent field capacity moisture such as would be found after heavy rainfall in areas with poor drainage. No indication were found of the influence of microorganisms or soil type on degradation.

Microbial Degradation. Probst et al. (1967) conducted experiments with trifluralin applied to autoclaved and nonautoclaved soils to determine the role of microorganisms in degradation. The experiment resulted in slightly more trifluralin degradation in the nonautoclaved soil; however, no specific soil organisms were isolated. No successful isolation of soil microorganisms has been indicated since then (Probst et al. 1975; Weed Science Society 1974).

Non-ionic Herbicides (Substituted Anilides)-Alachlor and Propachlor. Of the substituted anilide herbicides in common use, alachlor is the principle compound utilized within the Chesapeake Bay watershed. Alachlor was first introduced in 1969 for the control of annual grasses, certain broadleaf weeds and yellow nutsedge (Jaworski 1975; Weed Science Society 1974).

Alachlor is usually applied as a spray formulation or in granular form as a preemergent, early postemergent or incorporated into the soil prior to planting. Mode of action is primarily through absorption by germinating shoots or roots. Alachlor is readily translocated throughout the plant and functions as a protein inhibitor through the inhibition of GA-induced amylase production (Weed Science Society 1974; Devlin and Cunningham 1970 cited in Jaworski 1975).

Photochemical Decomposition. Weed Science Society (1974) described alachlor as having good resistance to decomposition by UV light. This would be expected under field conditions since alachlor does not absorb at wavelengths in excess of 2800 Å and sunlight is limited to above about 2900 Å. For this reason, Beestman and Deming (1974) concluded that photodecomposition along with chemical decompostion do not contribute in a significant way to losses of alachlor from soils. Crosby (1976) did not discuss alachlor specifically but referred to the level of photochemical knowledge of the amide or anilide herbicides as being "primitive".

Volatilization. Beestman and Deming (1974) studied volatilization of three acetanilide herbicides including alachlor. They found little volatilization under windy conditions from dry soils compared to the significant losses that occurred from wet, exposed soil.

Fate in Soil. Alachlor exhibits a moderate water solubility of 148 ppm at 20 to 25 C. Weber (1972) found that the effectiveness of alachlor was greater in soils with a high organic matter content compared to coarser textured soils. Stickler et al. (1969, cited in Weber 1972) determined that surface application compared to soil incorporation did not result in changes in alachlor effectiveness. Herbicide activity was not affected by soil moisture either.

Ballard and Santlemann (1973) studied alachlor activity in relation to clay, loam and sandy soils. Based on growth of wheat (Triticum vulgare) in the three general soil types, less growth inhibition was noted in soil with increasing sand or clay content compared to loamy soils. As adsorption decreased, so did alachlor activity.

Microbial Degradation. Microbial decomposition is the primary degradation mode for alachlor. Laboratory experiments performed by Beestman and Deming (1974) determined that alachlor and other anilide herbicides were 50 times more stable in sterilized soil then in viable soil.

Investigations by Tiedje and Hagedorn (1975) identified organic metabolities of alachlor when cultured with Chaetomium globosum. Chloride was one degradation product along with 2-chloro-2'6'-diethylacentanilide, 2,6-diethyl-N-(methoxymethyl) aniline; 2,6-diethylaniline; and 1-chloroacetyl-2,3-dihydro-7-ethylindole. Smith and Phillips (1975) found Rhizoctonia solani readily degraded alachlor when another carbon source was present.

Runoff Losses. Runoff loss of alachlor from silty clay loam soil of the Rhode River watershed area was monitored in 1976. The average leakage rate of alachlor was determined to be 0.02 percent (Wu et al. 1977).

Phenylurea Herbicides - Linuron. Herbicidal activity of the phenylureas was first described by Thompson et al. (1946, cited in Geissbuhler et al.1975). There are presently 20 to 25 commercially available urea herbicides of which linuron is the most common within the Chesapeake Bay area (Geissbuhler et al. 1975).

Linuron is normally sprayed on various important food crops for preemergent or postemergent control of broadleaf weeds and grasses. Plant absorption of linuron occurs primarily through the roots and secondarily by way of foliage and stems. Translocation occurs through the xylem with inhibition of the Hill reaction.

Photochemical and Chemical Degradation. Under hot, dry exposed conditions, linuron readily photodecomposes (Weed Science Society 1974; Crobsy 1976; Geissbuhler et al. 1963). Crosby (1976) cites Hill et al. (1955) as first discussing the photolysis of the substituted urea compounds. Although little has been published on the photochemical degradation products of linuron, information on similar urea herbicides is available (Weldon et al. 1969; Geissbuhler et al. 1963).

Urea herbicides in general are regarded as stable compounds under normal temperature and soil conditions (Geissbuhler et al. 1975). Hance (1967) used linuron in temperature experiments and concluded that losses of linuron from soils through chemical processes was unimportant.

Fate in Soil. Weber (1972) classified linuron as having moderate to low soil mobility that tended to decrease as soil organic matter increased. Generally, adsorption of linuron increased with increased clay organic matter content. Changes in phenylurea adsorption and mobility were not observed in varied pH solutions suggesting that the urea compounds did not ionize in solution (Hance 1969).

Leaching of linuron in two soil types was investigated by Dubey and Freeman (1965). Herbicide applications equivalent to 5.5 and 27.5 kg/ha tested leakage potential on Wheeling sandy loam and Maury silt loam in soil columns leached with 2.5 to 20 cm of water. Linuron was found to remain at or near the surface; however, the herbicide showed a greater potential for leaching in Wheeling sandy loam than in Maury silt loam. Similar findings by Upchurch and Pierce (1957) and Sherburne et al. (1956, both cited in Weber 1972) supported the marked mobility of the phenylureas in light soils. Linuron was found to have a water solubility of 75 ppm (Willis et al. 1975).

Linuron is considered to demonstrate low soil persistance (Scherer et al. 1963, cited in Geissbuhler et al. 1975) presumably due to the action of microorganisms. Willis et al.(1975) listed the soil half-life for linuron as approximately two months.

Microbial Degradation. Environmental conditions such as elevated temperature, high moisture and organic matter that favor microorganism growth also contribute to the deactivation of the urea herbicidal activity. Geissbuhler et al. (1975) cited extensive work by Borner et al. (1969) who used several urea herbicides including linuron with almost 100 fungal and bacterial species. The conclusion was reached that the ability of microbes to utilize the phenylureas was "rather widespread". Geissbuhler et al. (1975) maintained that assumptions as to microbial degradation capability based on the decomposition of similar urea compounds are not necessarily valid. The ability of a microbial species to degrade one type of phenylurea has not been shown experimentally to mean that the same microbe can degrade a closely related compound. Thus, the experiments by Hill et al. (1955) can not be taken as an indication that the strain of Pseudomonas that can degrade monuron will also be capable of utilizing linuron.

Runoff Losses. Loss of linuron applied at a rate of 2.24 kg/ha to soil of the lower Mississippi River Valley was monitored in a three year study by

Willis et al. (1975). This herbicide was applied to plots of Mhoon silty clay loam with 13 to 28 percent sand, 45 to 51 percent silt and graded to a 0.2 percent slope. Application resulted in maximum seasonal losses of 0.33, 0.27 and 0.04 percent, consecutively, for the three years. Higher losses to runoff were attributed to heavier rainfall events. Willis et al. suggested that with proper use linuron would not negatively impact adjacent waters.

Aquatic Weed Control. Lawrence (1968) studied the activity of linuron applied to submerged plants in growth pools. An initial application of 2.0 ppm linuron completely controlled naiad and pondweed for two years while common elodea showed approximately 50 percent control for the same time period. A combination of paraquat (0.1 ppm) and linuron (1.0 ppm) totally controlled naiad and elodea for two years. With the same combination, Lawrence found that 70 percent of the pondweeds were controlled the first year after application and 10 percent were controlled during the following year.

Recent field sampling by Correll et al. (1977) in several Chesapeake Bay estuaries (see following section) indicated that linuron had been found at concentrations up to almost 9 ppm in sediments. Bioassays by Correll et al. showed that 1 ppm linuron in sediments was sufficient to effect decreases in gross photosynthesis of *Zannichellia palustris*.

Herbicide Loadings in the Choptank Watershed

The Choptank River watershed was chosen in order to make some initial estimates of the possible impacts of herbicides on an important subestuary of the Chesapeake Bay. The Choptank River Basin is the second largest on the Eastern Shore, only the Pocomoke is larger. The Choptank River was chosen over the Pocomoke River as more typical of the Eastern Shore since the Choptank is surrounded by agricultural land. The Pocomoke River area is unique to the region because it is surrounded by a cypress swamp. Table 70 gives a brief comparison of the Choptank River characteristics with other important rivers in the Bay.

The Choptank has a diffuse source loading potential in the median range (see Table 70) of 1.3 (m^2 m^{-3}). Although this figure does not take into account different land use patterns or the relative flushing characteristics of each of the subestuaries, it seems to correlate well with patterns of occurrence of submerged aquatic plant species. This correlation is especially striking when the 1974 submerged aquatic plant occurrence (Kerwin et al. 1975) is plotted against the potential diffuse source loading potential for each estuary in Figure 41. The high loading ratios (> 5) of the Wicomico and Nanticoke Rivers may help explain why they have had no submerged aquatics in recent years. The influence in terms of land forcing functions is much greater in these areas. The Choptank ratio of 1.3 is much more in the middle range which makes it a more representative place to look at possible agro-chemical inputs on the Eastern Shore.

Acreage of the Choptank River drainage basin comprises nearly 58.7 percent of the acreage in the three Maryland counties (Talbot, Dorchester and Caroline) adjacent to the River. Table 71 gives approximate land use patterns in the Choptank watershed. Planting methods of corn, soybeans, wheat and barley largely determine the types and total quanities of herbicides potentially available for

Table 70. Physical characteristics of selected river systems in Maryland

	Total basin area[a] sq km	Navig. length[b] mi	Land MLW estuarine area sq km	Total MLW estuarine area 10^6 m^2	Total MLW volume 10^6 m^3	Intertidal estuarine volume 10^6 m^3	Potential diffuse loading ratio m^2 / m^3
Chester R.	1,036	38	834	202	835	81.7	1.0
Eastern R.	518	9	292	226	1045	74.4	.3
Choptank R.	2,246	58	1,880	366	1441	156.3	1.3
Nanticoke R.	2,111	34	2,036	75	190	52.9	10.7
Wicomico R.	619	10	572	47	107	134.3	5.3
Pocomoke R.	2,300	32	1,913	387	1073	252.0	1.8
Sassafrass	290	15	253	37	158	--	1.6
Patuxent R.	2,409	34	2,272	137	659	58.6	3.4
Potomac R.	8,664	98	7,413	1251	7288	341.7	1.0

[a] Maryland Department of Natural Resources files
[b] Cronin 1971

Table 71. Comparison of land use patterns in upper and lower Choptank River watershed areas

Land use	Upper Choptank[a] %	Lower Choptank[b] %
Forest	42	45
Cropland	35	28
Grassland	8	1
Orchard	-	2
Residential & commercial	8	2
Marshes (brackish)	-	22
Other (surface, mines, idle)	7	-
Total	100	100

[a] Fincher 1976

[b] Matthews 1963

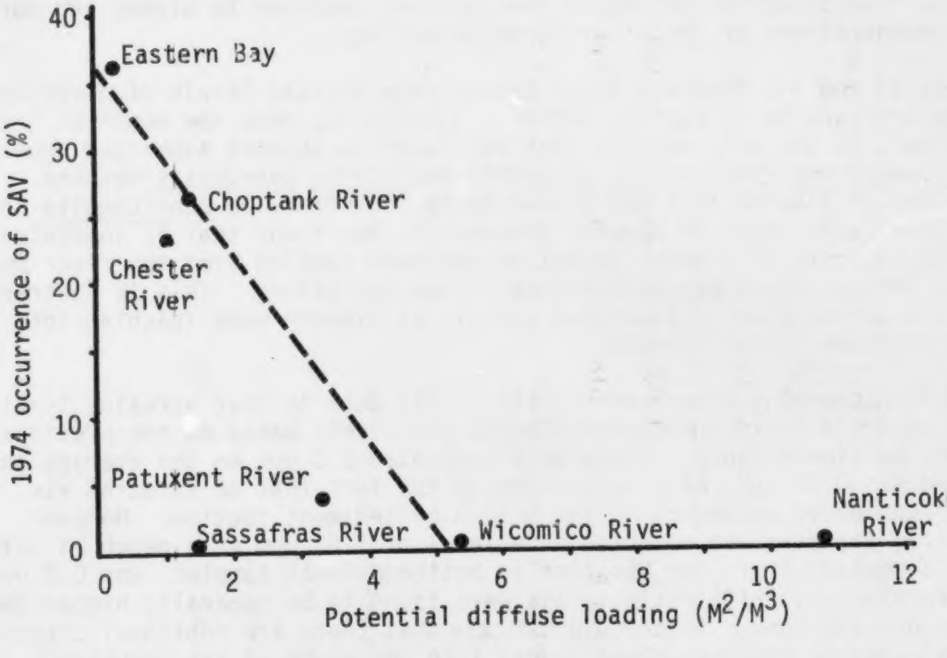


Figure 41. Submerged aquatic plant occurrence plotted against potential diffuse source loading ratio

leakage to receiving bodies of water. In the Choptank drainage basin, crops and respective planting methods are tabulated in Table 72 for the purpose of projecting the herbicide leakage potentials (see Table 73). The approximated 7.5 percent no-till corn acreage planted (Parochetti, personal communication) and 25.5 percent no-till soybeans (Maryland Department of Agriculture 1963-177, 1971-1977) have resulted in approximately 4,960 kg of paraquat applied to these crops.

Alachlor, atrazine and linuron are the most widely used herbicides in the Choptank area. A review of literature indicates that leakage figures for these herbicides could potentially be 13 kg alachlor, 538 kg atrazine and 105 kg linuron per year. If these loadings were diluted by the Choptank MLW volume and not incorporated into the sediments, we would expect to find 0.36 ppb atrazine, 0.07 ppb linuron and 0.0009 ppb alachlor. Less than a part per trillion of trifluralin would be expected. This is assuming no net transport out of (or into) the Choptank estuary. Although this latter assumption may be somewhat simplistic, it may actually represent an underestimation of herbicide loadings. In the Chester River there were researchers who found that more pollutants were carried upstream from the mouth on sediments coming from the Susquehanna River than were coming downstream.

Recent surveys of the Choptank River by the Chesapeake Bay Center for Environmental Studies (CBCES) have provided some initial data on herbicide levels and SAV populations at eight stations along the river from the mouth up to Denton (Correll et al. 1977). Atrazine, linuron, trifluralin and alachlor were analyzed from bottom sediments and filtered surface water. Alachlor was found in low concentrations at only a few stations compared to higher and more frequent concentrations of the other three herbicides.

Figures 42 and 43, Choptank River graph, show erratic levels of atrazine, linuron and trifluralin in surface waters. Station 72, near the mouth of the Choptank River, is the only station that was found to support submerged vegetation (Ruppia maritima Zannichellia palustris and Elodea canadensis despite concentrations of linuron that had proved to be inhibitory to Zannichellia in bioassays (see Table 74). Of special interest is the trend that is indicated in Figure 44. Levels of linuron in bottom sediment sampled from the river mouth upstream to Denton show a steady decrease in concentration. This is contrary to what would be expected if excessive amounts of linuron were leaching into the river from upstream soybean fields.

Another interesting aspect of Correll et al. data is that atrazine levels in sufrace waters (dissolved) exceeded expectation levels based on the previous calculations mentioned above. Field data indicated 2.0 ppb on the average and the expected was 0.36 ppb. Also surprising is the fact that no atrazine was reported in suspended sediments or the 0 to 3 cm sediment faction. However, at station 77, atrazine was detected at a level of 3.83 ppm at a depth of 3 to 6 cm. The detection limit for atrazine in bottom sediment samples was 0.2 ppm. Linuron, alachlor and trifluralin levels were found to be generally higher than expected by calculations. This could indicate that there are continual concentration build-ups or possibly other inputs into the mouth of the Choptank. It is also possible that actual leakage rates are much higher than expected or that CBCES has been identifying an artifact (Parochetti, personal communication) instead of an herbicide.

Table 72. Estimates of total amount of specific herbicides used for weed control in the Choptank drainage basin, 1975[a,b]

Cropping practice	Herbicide	Rate kg a.i./ha	Hectares treated	Herbicide kg a.i./ha
No-till corn	Paraquat	0.27	2,194	592
	Atrazine	1.37		3,005
	Simazine	1.37		3,005
	Paraquat	0.27	157	42
	Alachlor	2.20		345
	Atrazine	1.65		259
Conventional corn	Atrazine	2.20	7,249	15,947
	Atrazine	1.10	13,048	14,352
	Alachlor	1.65		21,529
	Atrazine	1.10	3,479	3,827
	Simazine	1.10		3,827
	Atrazine	1.10	870	957
	Simazine	1.65		1,436
	Atrazine	1.10	1,559	1,715
	Metolachlor	1.65		2,572
	Atrazine	1.10	1,450	1,595
	Butylate	3.30		4,785
	2,4-D	0.42	1,450	609
No-till soybeans	Paraquat	0.55	7,654	4,210
	Linuron	1.10	6,506	7,157
	Alachlor	1.65		10,735
	Linuron	0.55	383	211
	Oryzalin	0.55		211
	Linuron	0.83	612	508
	Miscellaneous	1.10	153	168
Concentional corn	Linuron	1.65	14,535	23,983
	Alachlor	1.65		23,983
	Linuron	0.83	2,236	1,856
	Trifluralin	0.83	2,684	2,228
	Napthalam[c]	1.10	1,118	1,230
	Dinoseb	1.65		1,840
	Alachlor	1.65		1,840
Conventional barley & wheat	2,4-D	0.42	3,687	1,549
	Dicamba	0.14	1,843	258

[a] Parochetti, personal communication

[b] Maryland Department of Agriculture files 1977

[c] Trade name

Table 73. Potential herbicide leakage, Choptank River drainage basin

Herbicide	Total grams applied	Possible percent leakage	Total herbicide leaked (grams)
Paraquat	4,958,140	a	a
Atrazine	44,795,974	1.20[b]	537,551
Simazine	8,452,791	a	a
2,4-D	2,939,251	0.21[c]	6,172
Dicamba	258,610	a	a
Trifluralin	2,260,228	0.05[d]	1,130
Linuron	34,836,516	0.30[d]	104,509
Alachlor	63,474,561	0.02[b]	12,694

[a] Data not available

[b] Wu et al. 1977

[c] White et al. 1976

[d] Willis et al. 1975

Table 74. Summary of bioassay results of various concentrations of atrazine and linuron on Zannichellia palustris[a]

Parameter measured	Herbicide concentration[b]	Bioassay results
Plant growth	1 ppm atrazine	increase
	1 ppm linuron	increase
Net O_2 production	1 ppm atrazine	increase until 20th day then decrease
	10 ppm atrazine	decrease
	100 ppm atrazine	decrease
	1 ppm linuron	increase until 12th day then decrease
	10 ppm linuron	decrease
	100 ppm linuron	increase
Gross photosynthesis	1 ppm atrazine	increase
	10 ppm atrazine	decrease
	100 ppm atrazine	decrease
	1 ppm linuron	decrease
	10 ppm linuron	decrease
	100 ppm linuron	decrease

[a] Correll et al. 1977

[b] Concentration for sediments. Water column (dissolved) concentrations were adjusted to ppb level

Further field work by Correll et al. has been performed on the Rhode and Severn Rivers and the Poplar Islands. Figures 42 and 43 show herbicide concentrations of dissolved and suspended solids of surface water. Linuron shows higher values at the river mouth (station 28) and atrazine and trifluralin are higher in Muddy Creek (stations 30.2 and 31.5) for the filtered surface waters in the Rhode River. This does not apply to suspended sediments concentrations which tend to be rather random.

Figure 44 depicts concentration levels found in bottom sediment samples. The two Muddy Creek stations exhibit the highest levels of all three herbicides. No strong correlations can be made between these concentrations and SAV populations. Ruppia and Zannichellia were noted throughout the five Rhode River stations despite what are considered to be inhibitory concentrations of linuron at most of the stations. Station 29 produced the highest amount of biomass (57,550 mg dry wt) yet showed almost 5 ppb linuron in suspended sediments and over 2 ppm in bottom sediments. Stations 31.5 and 28 ranked next in biomass (3,430 and 1,610 mg dry wt) despite higher herbicide concentrations than stations 28.4 and 30.2 which has lower plant biomass.

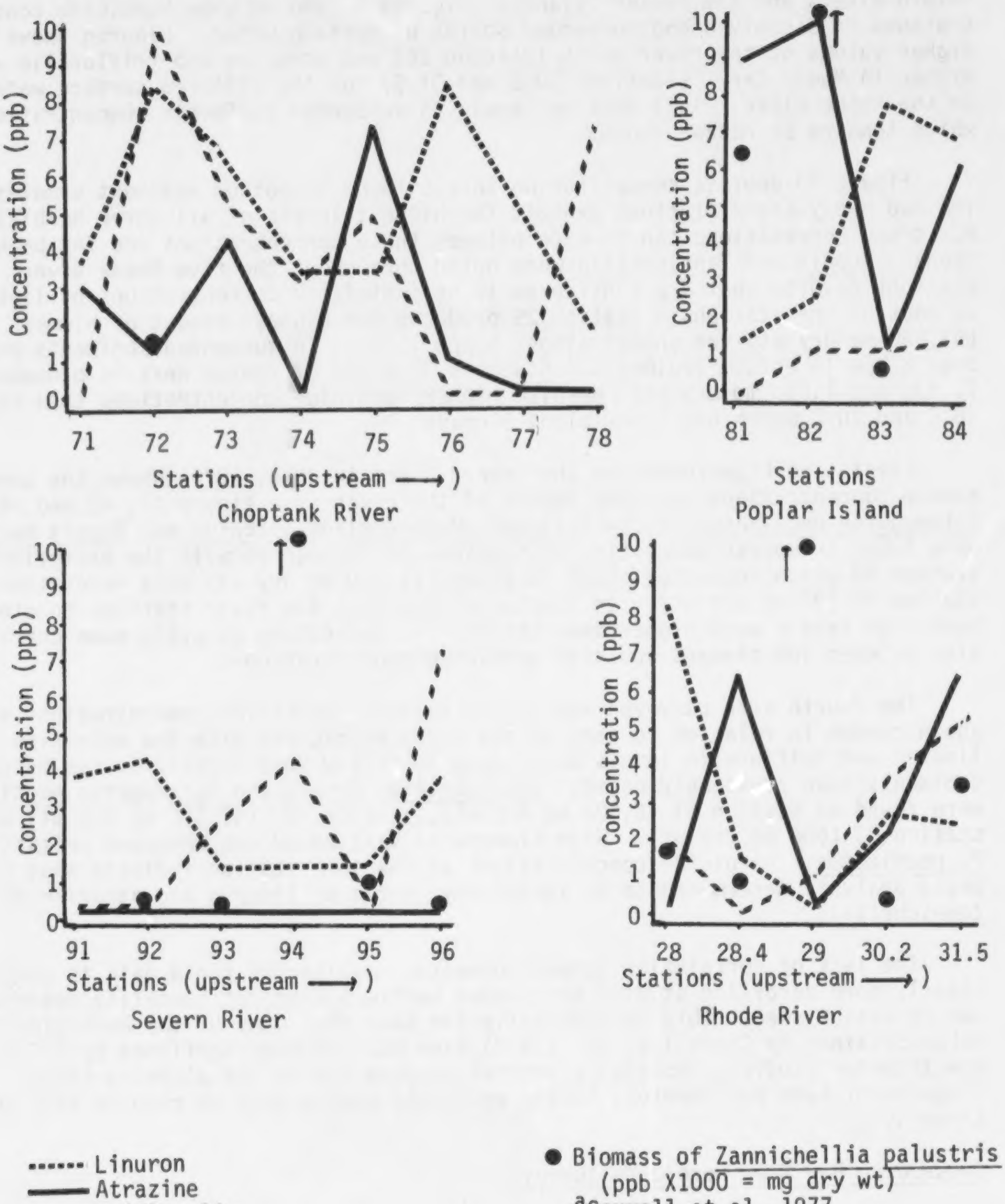
Similar work performed on the Severn River in June, 1976, shows the same random concentrations over the length of the river (see Figure 42, 43 and 44). Potamogeton pectinatus, P. perfoliatus, Myriophyllum spicatum and Ruppia maritima were found in sparse quantities at stations 92 through 96 with the exception of station 94 which supported about 30 times (11,410 mg dry wt) more vegetation than station 95 (20 mg dry wt). In looking at the last two river stations together, herbicide levels were higher than station 94, indicating possibly some correlation between low biomass and high herbicide concentrations.

The fourth area surveyed was Poplar Island. Herbicide concentrations are again random in relation to each of the three herbicides with the exception of linuron and trifluralin levels which seem to follow more closely in the bottom sediments than previously noted. Zannichellia, Ruppia and Potamogeton pectinatus were found at station 81 (6,500 mg dry wt), station 82 (42,150 mg dry wt) and station 83 (600 mg dry wt). High biomass at station 82 was composed primarily of P. pectinatus. Herbicide concentrations at the same station indicate what bioassay analysis determined to be inhibitory levels of linuron and atrazine to Zannichellia.

The lack of correlation between bioassay results and field data is confusing. Clearly more herbicide studies are needed before a clearcut causality pattern can be established. This is especially the case when many of the herbicide valuesobtained by Correll et al. (1977) have not yet been confirmed by G-C mass-spectrometer studies. Hopefully several studies now in the planning stages (Stevenson, Kemp and Boynton; Stotts and Orth) may be able to resolve this controversy.

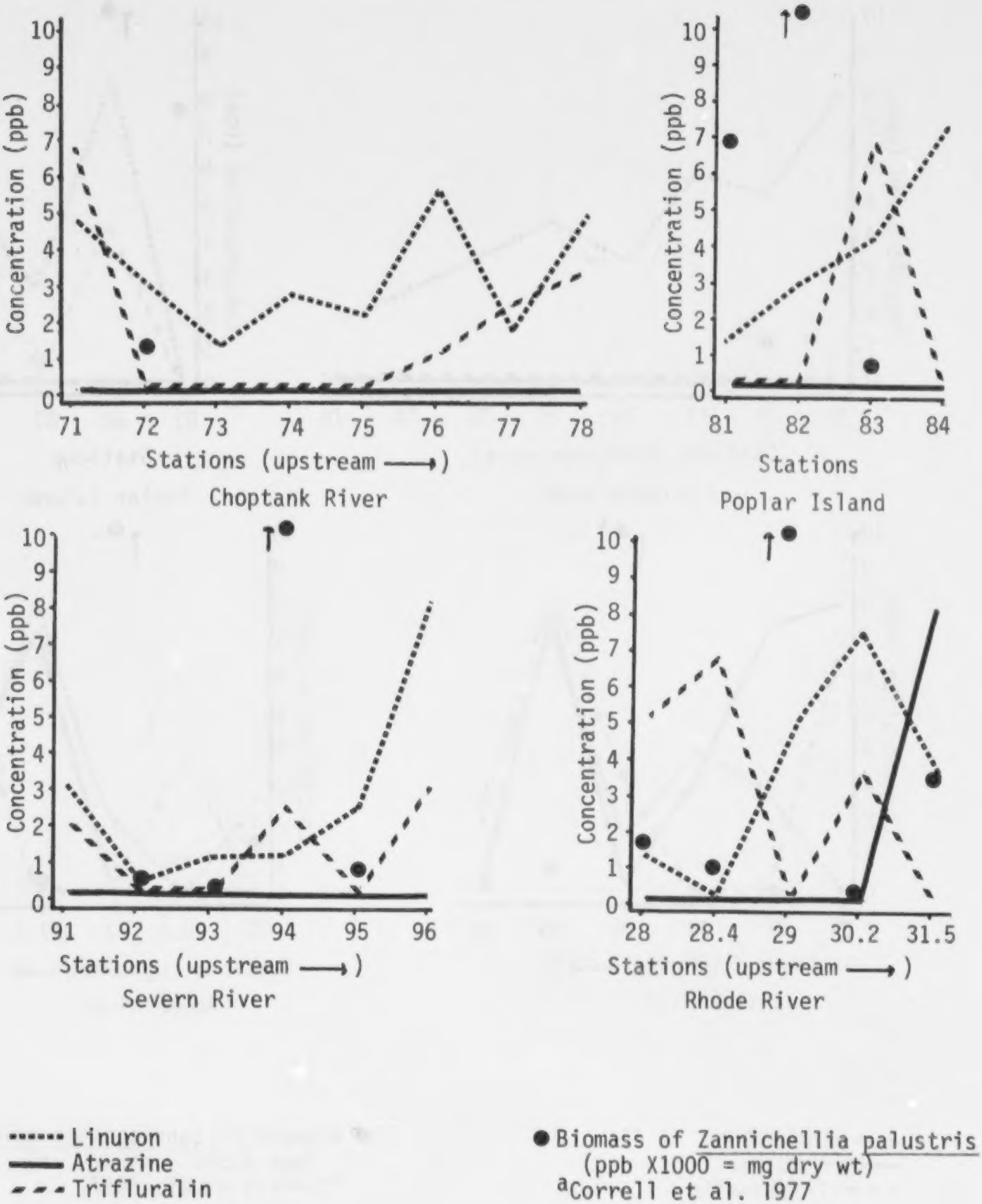
Chesapeake Bay Area Herbicide Survey

In recent years, herbicide usage has increased in the Chesapeake Bay area. In order to determine the extent of this increase, county agricultural extension agents from Maryland and Virginia were surveyed for information relating to local use of nine common herbicides. Tables 75 and 76 tabulate the data from the responding extension agents. These figures indicate a marked increase in total herbicide use for all responding counties from 1971 to 1975.



^{a}Correll et al. 1977

Figure 42. Analysis of surface water (dissolved) and biomass, June, 1976^a



[a] Correll et al. 1977

Figure 43. Analysis of surface water (suspended sediment) and biomass, June, 1976[a]

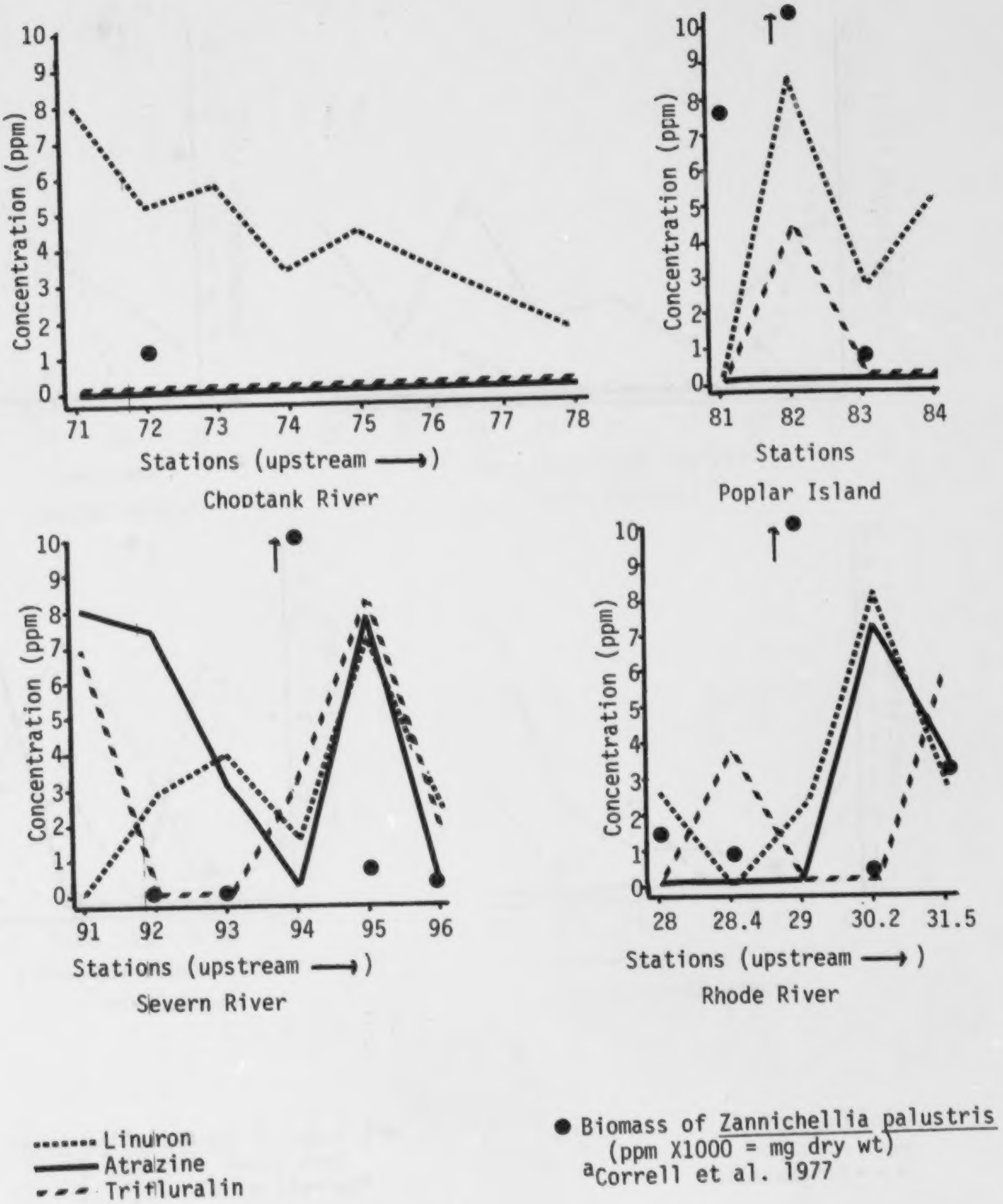


Figure 44. Analysis of bottom sediment (0 to 3 cm) and biomass, June, 1976[a]

	Cecil		Kent		Talbot		Caroline		Dorchester		Wicomico	
	1971	1975	1971	1975	1971	1975	1971	1975	1971	1975	1971	1975
Paraquat	454	726	170	2,268	*	4,082	1,134	3,093	1,134	1,701	227	603
Atrazine	15,422	*	31,752	45,360	40,370	44,226	13,347	12,474	31,752	24,948	12,248	5,443
Simazine	1,383	2,268	6,350	11,340	6,917	8,618	2,268	3,904	2,268	6,804	4,649	6,350
2, 4-D	907	771	765	454	6,124	2,381	1,650	1,633	4,536	1,814	1,447	2,004
Dicamba	136	181	57	454	2,041	454	550	273	454	454	842	1,489
Dinoseb	3,175	680	14,742	1,089	*	*	8,301	3,121	5,102	3,402	15,309	17,554
Trifluralin	2,722	680	680	2,722	907	680	1,701	1,252	2,041	3,402	1,429	1,361
Linuron	1,497	3,402	1,588	4,899	5,443	14,969	5,103	7,770	4,990	4,536	3,175	5,860
Alachlor	7,825	2,722	8,845	40,824	8,845	36,231	10,206	7,904	3,402	23,814	10,206	25,855
Total	33,521	11,430	64,949	109,410	70,647	111,641	44,260	42,234	55,679	70,879	49,532	66,519

	Baltimore		Anne Arundel		Charles		Calvert		St. Mary's		Totals	
	1971	1975	1971	1975	1971	1975	1971	1975	1971	1975	1971	1975
Paraquat	517	1,814	*	181	*	170	*	181	*	680	3,636	16,309
Atrazine	8,845	9,072	*	8,618	6,350	6,668	4,218	10,773	*	9,072	164,304	176,654
Simazine	5,103	6,804	*	*	454	2,268	295	599	*	5,443	29,687	54,398
2, 4-D	1,021	1,814	*	363	170	510	227	298	*	45	16,847	12,087
Dicamba	284	454	*	*	*	*	*	43	*	11	4,364	3,813
Dinoseb	5,783	850	*	*	*	272	137	181	*	91	52,549	27,240
Trifluralin	113	953	*	227	*	204	68	91	*	907	9,661	12,479
Linuron	204	408	*	204	340	816	771	2,699	*	8,165	23,111	53,728
Alachlor	1,225	2,041	*	567	*	2,722	771	2,699	*	5,443	51,328	150,822
Total	23,095	24,210	*	10,106	7,314	13,630	6,487	17,564	*	29,857	355,484	507,530

[a] Virginia Polytechnic Institute files 1977

*data not available

Table 76. Estimate of the use of selected herbicides (kg a.i.) by county in Virginia, 1971 and 1975[a]

	Accomack		Northampton		Pr. William		King George		Westmorland		N. Cumberland		Lancaster		Richmond	
	1971	1975	1971	1975	1971	1975	1971	1975	1971	1975	1971	1975	1971	1975	1971	1975
Paraquat	*	449	*	510	91	*	11	391	91	1,030	340	795	11	*	236	*
Atrazine	*	4,168	*	1,143	581	*	2,023	3,792	4,419	8,446	4,536	7,276	3,489	*	2,522	*
Simazine	*	680	*	136	*	*	*	1,847	299	900	41	2,803	*	*	27	*
2,4-D	*	2,427	*	1,542	3,459	*	1,854	386	272	425	1,012	251	370	*	1,474	*
Dicamba	*	227	*	397	68	*	204	45	23	63	*	*	*	*	*	*
Dinoseb	*	*	*	510	*	*	816	68	2,381	1,134	181	81	1,270	*	*	*
Trifluralin	*	1,157	*	885	*	*	41	136	84	1,433	163	3,147	136	*	231	*
Linuron	*	1,012	*	1,111	*	*	113	599	649	2,368	975	1,565	48	*	771	*
Alachlor	*	3,289	*	408	159	*	*	1,792	363	4,253	1,814	2,211	91	*	91	*
Total	*	13,409	*	6,642	4,358	*	5,062	9,056	8,581	20,052	9,062	18,129	5,415	*	5,352	*

	Middlesex		Gloucester		King & Queen		New Kent		York		Isle of Wight		Suffolk		Chesapeake		Totals	
	1971	1975	1971	1975	1971	1975	1971	1975	1971	1975	1971	1975	1971	1975	1971	1975	1971	1975
Paraquat	6	850	68	284	*	340	*	499	6	*	*	102	284	*	*	367	1,144	5,617
Atrazine	2,085	5,117	6,174	7,802	3,647	4,763	*	3,409	102	*	10,705	10,977	5,833	*	*	3,958	46,116	60,851
Simazine	*	2,068	68	2,268	*	218	*	1,524	*	*	54	109	45	*	*	*	534	12,553
2,4-D	261	567	45	45	454	317	*	159	145	68	1,089	2,336	1,270	*	*	2,495	11,705	11,018
Dicamba	13	51	5	5	23	7	*	86	23	*	*	*	*	*	*	*	359	881
Dinoseb	1,814	204	181	181	*	680	*	544	*	*	*	12,882	4,082	*	*	1,021	10,725	17,305
Trifluralin	*	361	204	2,858	*	293	*	136	*	*	27	113	54	*	*	*	940	10,519
Linuron	844	1,894	132	2,268	*	680	*	635	23	39	34	*	567	*	*	1,792	4,156	13,963
Alachlor	*	794	1,633	10,206	454	3,062	*	2,199	*	*	728	24,585	3,719	*	*	9,952	9,052	62,751
Total	5,023	11,906	8,510	25,917	4,578	10,360	*	9,191	299	107	12,637	51,104	15,854	*	*	19,585	84,731	195,458

[a] Virginia Polytechnic Institute files 1977

* data not available

CHLORINE

Since the early 19th century, chlorine has been utilized as a sanitizing agent. By 1912, it was widely used for the destruction of bacteria in both sewage and potable waters (Schultze 1974). Chlorine has been found to be effective for:

. disinfection of clams and oysters prior to marketing

. control of midge larvae and alage in swimming pools

. controlling taste and ordor in municipal water supplies

. sterilization of fish hatchery water supplies

. reduction in biochemical oxygen demand of domestic wastewater

. odor and fly control in wastewater treatment plants

. industrial wastes

. antifouling in heat exchangers and piping of water cooling systems

. pulp and paper production (bleach)

. herbicide and pesticide production

The ecological impact of this strong oxidant and its by-products on the marine environment appears to be vast. It was vertified several years ago that chlorinated hydrocarbons such as DDT and DDE, dieldrin and PCBs which enter seawaters from land runoff, sewage outfalls and the atmosphere were extremely dangerous to marine flora and fauna (Goldberg et al. 1971).

Goldberg et al. (1971) cited the work of Nimmo et al. (1970) who found that continuous exposure of 0.2 ppb chlorinated hydrocarbon killed shrimp. They also refer to Duke et al. (1970) who determined that one ppb of the PCB Arcolor 1254 caused a 20 percent decrease in oyster shell growth. Duke corrolated HCl residues in mollusks to nearby agricultural chemical applications.

Fish and birds have also exhibited negative effects due to chlorinated compounds. Concentrations of chlorine in lipid ovary tissues affected new born growth. A 5 ppm chlorine residual caused developmental failure of sac drys containing young fish (Goldberg et al. 1972). Deaths of bald eagles (Mulhern et al. 1970), common loon (Butler 1966) and peregrine falcons (Jeffries and Prestt 1966) are caused by chlorinated hydrocarbons through eggshell thinning (Goldsberg et al. 1971). Bald eagles and peregrine falcons are included on the Department of the Interior's endangered species list.

Aquatic research has demonstrated that phytoplankton show decreased growth, developmental failures and increased mortality rates in response to chlorine. Chlorine inhibits photosynthesis in single-celled marine plants causing these effects. Particularly important is that these plants may be a vehicle for transferring these potentially toxic compounds to higher trophic levels (Goldberg et al. 1971).

In bioassay tanks using estuarine waters, Wester and Rawles (1976) described symptoms in aquatic macrophytes of growth retardation, loss of chlorophyll and collapse when total available chlorine levels ranged from 0.05 to 0.125 ppm. "Field observations, plot studies and data from controlled environment investigations with Caboma caroliniana, Elodea canadensis, Potamogeton crispus, P. pectinatus and Vallisneria spiralis indicate that chlorine pollution may be a significant cause of this critical environmental problem affecting these once locally prevalent plants"

The fate of chlorine in estuarine ecosystems is largely unknown at the present time and laboratory chemistry techniques for chlorine are not dependable yet (Davis, personal communication). Experimental data does show that chlorine acts differently in marine waters compared to freshwaters. The 60 ppm of bromide that characterizes marine waters is readily oxidized by the introduction of strong oxidants. In marine waters, the resulting compounds are brominated while in freshwater, analagous chlorinated compounds are formed (Sugam and Helz 1977; Davis et al. 1977). An estimate of the possible number of theoretical halogenated inorganic and organic by-products produced is 1,500 (Davis, personal communication). The measurement of these compounds requires an intensive inventory and analytical effort. The common technique for measuring the active oxidative stage or residual chlorine is the amperometric titrator method. This technique does not accurately record hypobromite and bromate or give any clue to the number of halogenated by-products (Davis et al. 1977).

In the Chesapeake Bay, chlorine and chlorine by-products enter the estuary via sewage treatment plants, water treatment and cooling waters from electric power plants, run-off from agricultural pesticide application and industrial effluents. The amount of chlorine used for sewage treatment has been estimated as ten times the amount used as biocide for power production cooling waters (Block et al. 1977). Based on 1973 data available from Martin Marietta Corporation (Polgar, personal communication), it has been estimated that chlorine used in municipal treatment is 12,200,000 kg and 7,000,000 kg is applied by power generating facilities. Jolley, in a 1975 symposium on the environmental impacts of water chlorination, estimated that at least 3 percent of this input could produce halogenated organic species of a persistent nature (Davis and Middaugh 1975).

A survey of chlorine usage from sewage treatment plants in the Chesapeake Bay shows an increase greater than 2.5 times from 1971 to 1975 (see Table 77) in each of four river systems that were analyzed.

There is currently no direct measurable evidence relating the Baywide decline in submerged aquatic vegetation to levels of chlorination. Bay vegetation was at its lowest point in 1975 but has increased in subsequent years. This does not clearly correlate with chlorine inputs which have increased substantially every year in the 1970s due to increased use by sewage treatment plants and power plants. Lower rainfall in 1976 and 1977 provided less potential fresh water dilution of the increasing chlorine load. Therefore, if chlorine alone is suspect as the agent causing Baywide declines of aquatic vegetation, we would expect to see a continued decline in SAV after 1975. This has not been found to be the case. Therefore, either chlorine is not at fault or it is acting synergistically with other factors.

However, in areas of the Bay which have high concentrations of both power plants, which use chlorine in large pulses to reduce fouling organisms, and STPs, there is a likelihood that chlorine may be a significant factor. The Department of the Interior, National Ecological Services Lab has a continuing monitoring and chlorine bioassay program which should provide a definitive answer concerning the extent of chlorine impacts on SAV in the future (Wester, personal communication). Therefore, although chlorine may be an important factor in causing SAV declines in population impacted watersheds of the Western Shore, there is little evidence presently linking it as a prime factor to the Baywide submerged aquatic decline. Any impact to SAV from chlorine can only be speculative at this point. However, this conclusion must be considered as tentative until a better working knowledge of the complete chemistry of chlorine is available for the Chesapeake Bay.

Table 77. Chlorine usage in four major rivers of the Chesapeake Bay estuary, 1971 and 1975[a]

River	1971	1975
Patuxent River	5,100	15,125
Pocomoke River	350	700
Chester River	140	550
Choptank River	1,500	3,600
TOTAL	70,090	19,975

[a] Maryland Department of Health and Mental Hygiene, Environmental Health Administration files 1977.

TURBIDITY

Turbidity refers to a variety of water quality parameters which include suspended organic and inorganic particulates, coloring or staining from dissolved organic matter and plankton. These varying components of turbidity can affect submerged aquatics in different ways. Particulates can physically block the penetration of light through the water column. Stained or colored waters diffently absorb various wavelengths of sunlight. Plankton can cause scums, mats or blooms which physically block sunlight and directly utlize red and blue wavelengths for photosynthesis. The dissolved and particulate matter entering the water column can serve as a vehicle for the introduction of soluble pollutants into an estuary. Whether these pollutants are heavy metals, greasy wastes, agro-chemicals, excessive nutrients, etc., they can be diluted, absorbed into substrate material, oxidized or precipitated.

Turbidity not only varies as to components but has high seasonal variability. Winter usually is the period of lowest turbidity. The spring rains increase suspended solids loading, and warming temperatures in summer promote plankton blooms. In addition, storms throughout the year can resuspend bottom sediments causing short periods of high turbidity.

Sources of suspended particulates are categorized by Biggs (1970) as: external, from freshwater runoff; marginal, from shoreline erosion; and internal, from biological production. In studying the Chesapeake Bay, Biggs estimated a yearly suspended solids input based on the year February 1, 1966 to January 31, 1967. The net volume of introduced sediments for the upper Bay (above Annapolis) was estimated as 1.6 x $10^6 m^3$ and 0.8 x $10^6 m^3$ for the central Bay (from Annapolis to a point just south of the Patuxent River mouth). With uniform distribution, the nominal rates of sedimentation would be 3.7 mm/year for the upper Bay and 1.1 mm/year for the central Bay.

The impact of suspended solids on SAV is not wholly negative. The deposition of silt and clay particles can aid in building up suitable bottom substrates in barren areas or add nutrients to existing substrates (Odum and Wilson 1962). Submerged aquatic vegetation, in turn, aids in the settling out of suspended solids.

Turbidity has often been implicated as a prime cause for the decline of submerged vegetation in the Chesapeake Bay (Martin and Uhler 1939; Bayley et al. in press; Rawls et al. 1975). However, the Bay has always been turbid from natural causes, due to the dynamic nature of a wind driven estuary. Many people knowledgeable about the Bay feel that turbidities have increased dramatically over the last forty years. Logically this would seem to be valid. Increased boat traffic and shoreline construction alone would appear to have augmented turbidity levels. However, the data available from the Department of Natural Resouces show that yearly averages of suspended solids in the Maryland portion of the Bay have declined in the 1970s (see Table 78). Furthermore, there is a positive correlation (r=.58) of SAV overall occurrences and yearly average turbiditity levels. This is in the opposite direction expected, assuming that high sediment loadings and resulting turbidities inhibit plant growth. Focusing on the turbidities during the growing season (April to September) (see Table 78), there is a little more reasonable trend.

The year of highest suspended solids during the growing season was 1975, also the year of lowest submerged aquatic vegetation occurrence. However, this does little to demonstrate that a clear relationship exists between growing season suspended solids and SAV declines, This is because the average suspended solids for the growing seasons of 1967 to 1969 was 54.3 mg/l compared to 23.8 mg/l recorded in the 1970s. This decrease occurred during a period when SAVs were declining. This second positive correlation adds to the doubt that SAV disappearances can be attributed alone to increasing turbidity effects due to increasing levels of suspended sediment.

Table 78. Yearly averages of suspended solids (mg/l), Maryland Chesapeake Bay, 1971-1976[a]

Year	Suspended solids (mg/l)				
	Apr/May	June/Jly	Aug/Sept	Apr/Sept	Average
1971	32.0	26.3	14.4	24.2	30.1
1972	37.8	-	5.5	21.6	32.9
1973	9.0	7.0	6.8	7.6	7.6
1974	25.8	40.3	26.5	30.9	29.4
1975	36.5	28.7	59.2	41.5	27.4
1976	12.0	23.7	15.9	17.2	21.7

[a] Maryland Department of Water Resources files 1977

Francis Uhler (Patuxent Wildlife Station, U.S. Fish and Wildlife Service) and Vernon Stotts (Maryland Wildlife Administration) maintain that the absence of submerged aquatics in an area such as Nanticoke River is due to higher turbidities (personal communications). However, Maryland Department of Natural Resources water quality data show that in the period 1966 to 1976 the Nanticoke actually had lower suspended solids (33.9 mg/l) than the Choptank (36.6 mg/l) or Chester (41.5 mg/l). Also, in most tributaries in the Bay, the suspended solids were lower in the 1970s than in the 1960s. This again suggests that it is difficult to attribute submerged aquatic declines in the 1970s to turbidity alone (see Figure 45).

Experimental evidence relating directly to the effects of Chesapeake Bay turbidity on SAV is scarce, at least until recently. Field surveys conducted by the Chesapeake Bay Center for Environmental Studies (Smithsonian Institution) in the Rhode and Severn Rivers do not indicate a strong correlation between turbidity and SAV occurrence. In both rivers, the greatest abundance of SAV was found in the areas of maximum turbidity where the fresh and brackish waters meet. However, in the Choptank River, vegetation was not found above Cambridge where the zone of maximum turbidity is located.

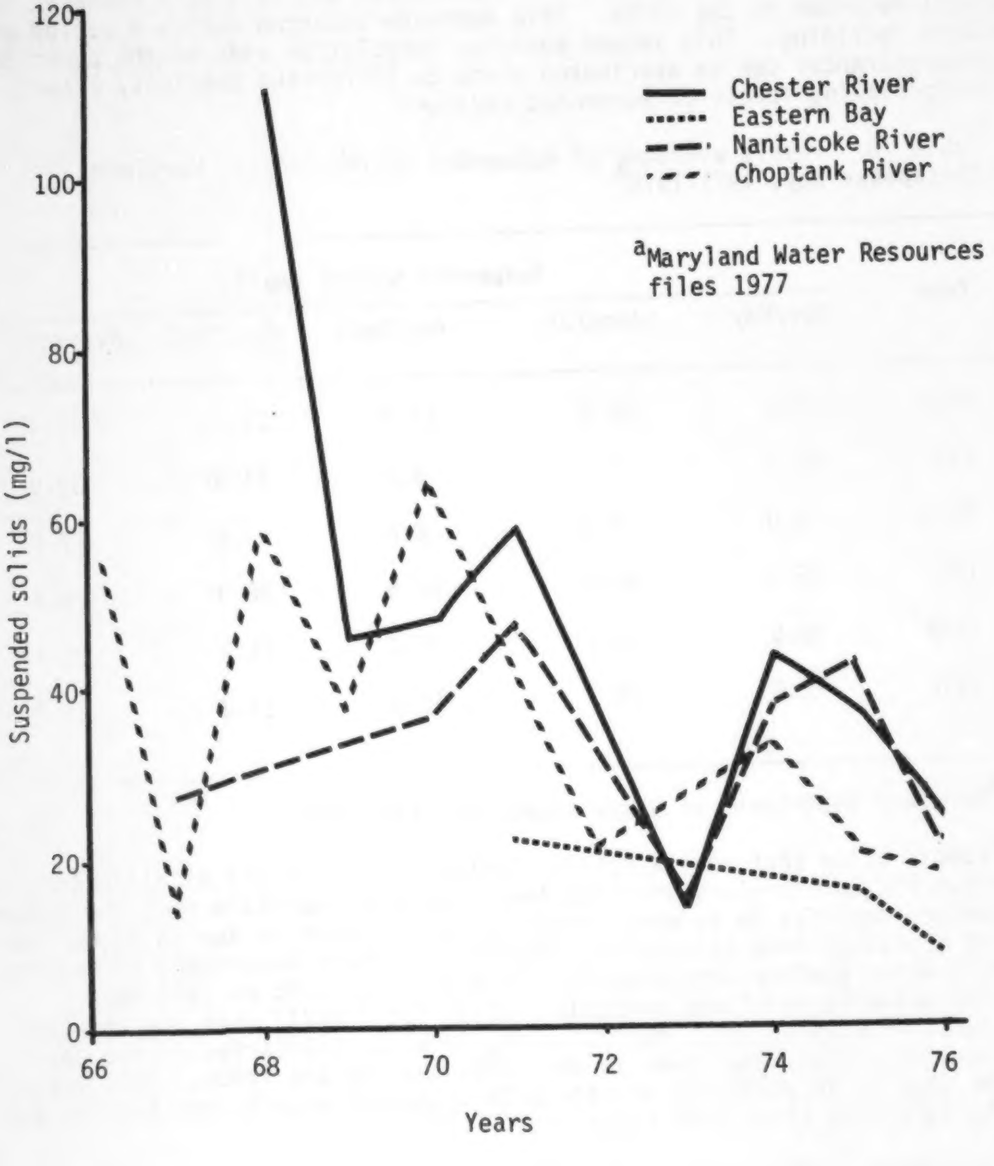


Figure 45. Suspended sediment sampling data (yearly averages) for four areas in the Chesapeake Bay[a]

Other investigations into the effects of turbidity on SAV include field determinations and laboratory experiments performed by Bourn (1932) that were determined to show conclusively that turbidity was probably the chief factor responsible for SAV destruction in Back Bay and Currituck Sound. Turbidity was determined to be deterimental due to the lessening of light penetration and silting of plant leaves.

Hurricane Agnes in 1972 provided a unique and recent opportunity to study the Bay's recovery time subsequent to a major storm. The distribution of suspended solids was greatest in the upper Bay and persisted there longer than in the lower Bay. By the end of July, the total distribution of suspended solids in the upper Bay was near normal for that time of year (Davis 1974). Besides sampling for SAV, the MBHRL Survey has measured turbidity with a Secchi disk from 1972 through 1976 (see Table 79). Data from 1972 was taken in July and August and supports Davis' findings that the Chesapeake Bay recovered within at least two months. Average Secchi disk readings for 1972 are generally lower than the readings taken in 1973 through 1976. Sampling averages for the entire Maryland Bay indicate a slight increase in light penetration from 1972 to 1976 with highs in 1974 and 1975. These measurements are taken within a two month period in the summer and so do not reflect yearly averages.

LIGHT

Light transmission is similarly affected by both turbidity and color. Waters high in turbidity or coloring matter are usually penetrated more deeply by red light (600+ mμ). Blue light (400 to 600 mμ) was found to penetrate farther in waters with lower coloring and turbidity (Blackburn et al. 1968).

Light can be a limiting factor to SAV by determining the extent of the photic zone. Sculthorpe (1967) discussed this relationship and stated that SAV can inhabit suitable areas with as little as 1 to 4 percent of the average surface light intensity. Light can also have an effect on seed germination (Hutchinson 1975). Potamogeton spp. tend to require light for proper seed germination, while light tends to be inhibitory to Najas spp. seeds.

C.H. Southwick and F. Pine at Johns Hopkins University looked at the possibility of seed germination being negatively impacted by changes in the incidence of sunlight reaching the water surface. A March, April, May and June average was determined for 1968 through 1973 based on National Oceanic and Atmospheric Administration figures taken at Baltimore-Washington Airport. These months were chosen to correspond with normal Bay seed germination times. Figures for the percent of total possible sunlight are in Table 80. Percentage figures through 1973 showed a very pronounced decrease in percent sunlight for the spring germination season. However, data for 1974 through 1976 reverses this percentage decline, especially for the year 1976. It is not improbable that declines of some species of SAV may be related to fluctuations in sunlight and seed germination potential. Further experimental studies into the relationships of light and turbidity in SAV microcosms would be helpful in

Table 79. Average Secchi disk data (cm) by river system, Maryland Chesapeake Bay, 1972-1976[a]

Area code	River system	1972	1973	1974	1975	1976
1	Elk and Bohemia Rivers	33.0	35.1	-	25.7	36.3
2	Sassafras River	34.3	52.3	-	29.2	51.1
3	Howell and Swan Points	33.8	75.4	-	61.2	57.7
4	Eastern Bay	67.3	62.5	76.5	54.6	75.9
5	Choptank River	60.7	62.5	84.3	61.5	64.3
6	Little Choptank River	64.5	59.4	66.8	63.8	78.5
7	James Island and Honga River	70.1	64.0	74.2	67.1	73.4
8	Honga River	78.2	67.3	72.6	68.8	67.8
9	Bloodsworth Island	73.7	87.6	94.7	177.0	83.3
10	Susquehanna Flats	64.5	65.5	82.6	33.8	76.5
11	Fishing Bay	49.5	77.0	85.6	75.7	54.1
12	Nanticoke and Wicomico Rivers	55.4	58.9	65.8	61.0	58.9
13	Manokin River	94.2	94.7	101.3	107.4	81.0
14	Patapsco River	73.7	80.0	67.8	-	70.1
15	Big and Little Annemessex Rivers	109.7	92.7	96.3	88.1	85.1
16	Gunpowder and Bush River Headwaters	42.9	38.3	46.7	-	53.8
17	Pocomoke Sound, Maryland	101.6	82.0	-	96.8	85.9
18	Magothy River	83.8	97.3	73.4	-	74.4
19	Severn River	97.3	70.4	79.5	-	86.4
20	Patuxent River	80.3	80.8	61.5	66.8	62.7
21	Back, Middle and Gunpowder Rivers	79.5	75.7	73.2	75.4	61.2
22	Curtis and Cove Point	45.2	77.0	81.8	58.9	73.7
23	South, West and Rhode Rivers	74.7	66.0	61.2	48.5	67.1
24	Chester River	76.2	73.4	100.1	87.9	85.1
25	Love and Kent Points	89.7	74.7	117.6	72.1	89.9
26	Smith Island, Maryland	78.5	76.2	89.7	139.4	87.6
	AVERAGE	70.1	71.1	79.5	76.2	71.4

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[a] U.S. Fish and Wildlife Migratory Bird and Habitat Research Laboratory files 1977

evaluating their importance in Baywide declines. Especially important is the manipulation of radiant energy inputs with other stress factors (i.e. herbicides, etc.) to determine if there are synergistic effects occurring.

Table 80. Percent of total possible sunlight reaching the surface, Baltimore-Washington International Airport[a]

Year	Spring mean[b]	Yearly mean
1968	59.25	58
1969	59.75	54
1970	50.50	56
1971	58.75	55
1972	52.25	52
1973	48.00	54
1974	56.00	55
1975	54.75	53
1976	63.50	62

[a] National Oceanic and Atmospheric Administration, Ashville, South Carolina, personal communication

[b] March, April, May and June

SALINITY

The Chesapeake Bay can be partitioned according to four major salinity zones: freshwater (0 to 0.5 ppt); oliogohaline (0.5 to 5 ppt); mesohaline (5 to 18 ppt); and polyhaline (18 to 25 ppt). The polyhaline regime is mainly restricted to the Virginia portion of the Bay whereas the mesohaline zone dominates the Maryland Bay area. Freshwater is found in all of the Bay subestuaries along with oliogohaline and mesohaline conditions.

Species of submerged aquatic vegetation tend to be distributed within the Bay according to this salinity regime. General tolerance levels for Bay vegetation according to Steenis (1970) are as follows:

3 ppt
Najas guadalupensis (southern naiad)

3-5 ppt
Chara spp. (muskgrass)
Vallisneria americana (wildcelery)

12-13 ppt
Elodea canadensis (elodea)
Myriophyllum spicatum (Eurasian watermilfoil)
Ceratophyllum demersum (coontail)

20-25 ppt
Potamogeton perfoliatus (redhead grass)
P. pectinatus (sago pondweed)
Zannichellia palustris (horned pondweed)

over 30 ppt
Ruppia maritima (widgeongrass)
Zostera marina (eelgrass)

These tolerance ranges tend to be both narrow and high compared to other experimental evidence. However, they do provide adequate information as to species that tend to inhabit similar ranges. Thus eelgrass dominates the lower mesohaline and polyhaline areas in the Virginia and southern Maryland Bay portions. Widgeongrass, due to its extremely wide salinity tolerance is codominant with eelgrass but is also found throughout most areas in the Bay. Pondweeds are found in fresh, oligohaline and upper mesohaline portions of Maryland and in the upper and middle reaches of Virginia rivers.

Increases in salt content generally result in an overall growth reduction. Due to increases is osmotic pressure resulting from a higher salt content, the plant is required to spend energy in salt absorption rather than in growth. A further reduction in growth rate results from the effects of sodium on the calcium regime and related cell wall structure (Chapman 1960).

Teeter (1965) studied the effects of sodium chloride on Potamogeton pectinatus seed germination and growth plus tuber production and growth. As the NaCl was increased to 12 ppt the plants showed visible damage; they were small and bushy with short, stubby blue-green leaves, short roots and the root caps and distal portions of the roots turned brown. Seed production was reduced 50 percent at 2 to 2.5 ppt. Tuber production was reduced 50 percent below 1 ppt and above 7.5 ppt, however at 3 ppt there were more tubers with greater weight produced. Reduction in vegetative growth occurred at 9.5 to 11 ppt.

Teeter found a 50 percent reduction in seed germination at 6 ppt. At 15 ppt, sago pondweed seeds were retarded by 19 days but recovered and germinated in 7 to 8 days after being placed in tapwater. These germination results after transfer from saline to freshwater confirmed Chapman's (1960) earlier work. Chapman had determined that seed germination of saltwater plants occurred optimally in freshwater because saltwater inhibited water uptake by the seeds.

Ecological and physiological studies by Bourn (1932) included data on the growth and salinity relationship of *Potamogeton pectinatus*, *P. perfoliatus* and *Ceratophyllum demersum*. *P. pectinatus* continued to grow well in salinities up to 12 ppt. Abundant roots developed, seed production was normal and a normal green color persisted. *P. perfoliatus* produced similar results up to 11 ppt. From 11 to 12 ppt there was marked reduction in leaf size, and plants grew spindly. *C. demersum* decreased in growth in proportion to increases in sea water concentration. Normal development continued up to 6.6 ppt, but this ceased in concentrations of 6.6 ppt or greater. At 6.6 ppt leaves were reduced in size and exhibited curling tendencies and stems were spindly The limit of endurance was 8 ppt. Death and disintegration in one week followed concentrations higher that 8 ppt.

Photosynthesis and respiration in relation to salinity were studied by McGahee and Davis (1971). *Myriophyllum spicatum* was utilized. It was found that a 4 to 8 ppt saline solution enhanced photosynthesis while at 13 to 14 ppt where *Myriophyllum* grew naturally, photosynthetic rates were good, but not as high as those at 4 to 8 ppt. Deletrious effects became apparent at 16 ppt and gradual increases to 32 ppt revealed drastically lowered photosynthetic rates. Respiration at all tested salinity concentrations showed no effects.

Haller et al. (1974) studied the effects of salinity on several species of SAV to determine toxicity levels. Results indicated that 6.66 ppt or higher was toxic to *Vallisneria americana*, 10.0 ppt and over toxic to *Najas guadalupensis* and salinities of 13.32 and more toxic to *Myriophyllum spicatum.*

There has been speculation that rapid salinity changes in the Bay might be a factor in SAV declines.Elser (1969) refers to increasing salinity levels from 1962 through 1966. Average salinity data by river area obtained from the MBHRL Survey files indicates that from 1971 through 1976 salinity has decreased in the Bay (see Table 81). Because the MBHRL Survey stations are sampled once a year in the summer, other salinity data was obtained from the Chesapeake Biological Laboratory. This is presented in Table 82 and consists of mid-monthly readings of salinity from the Solomon's pier and shows again generally decreasing salinities in the 1970s.

Cronin (1976, p. 6) suggests that low salinity or a related factor"... has apparently been the primary cause of large (SAV) losses."

However, as cited above, most species of SAV (and most halophytes) have little or no problem in tolerating decreased salinity levels. In fact, with a reduction in salinity, enhancement of growth and germination often occurs. Therefore the assertion that the decline of the SAV in the 1970s is due to lowered salinities commonly associated with Hurricane Agnes is difficult to substantiate with scientific evidence. Any effect of Hurricane Agnes may have occurred through sediment loading problems already discussed or other toxicity problems discussed below, but probably not salinities.

Table 81. Average salinity (ppt) by river system, Maryland Chesapeake Bay, 1971-1976[a]

Area code	River system	1971	1972	1973	1974	1975	1976
1	Elk and Bohemia Rivers .	-	1.000	1.000	1.000	3.525	1.000
2	Sassafrass River	-	1.000	1.000	2.570	4.430	-
3	Howell and Swan Points	4.200	1.000	1.750	6.400	4.967	2.960
4	Eastern Bay	17.181	9.242	10.038	15.238	10.252	7.100
5	Choptank River	15.598	9.567	9.350	14.146	9.259	6.727
6	Little Choptank River	18.932	12.468	11.726	16.422	12.658	7.895
7	James Island and Honga River	20.135	15.103	13.794	18.200	15.694	9.874
8	Honga River	19.943	15.503	14.320	18.723	16.200	9.038
9	Bloodsworth Island	21.217	17.395	18.933	21.381	12.374	13.909
10	Susquehanna Flats	1.500	1.000	1.000	1.000	3.856	1.000
11	Fishing Bay	18.164	14.468	14.616	17.656	15.040	13.364
12	Nanticoke and Wicomico Rivers	15.400	11.213	11.203	14.026	13.897	10.453
13	Manokin River	22.620	17.540	16.760	20.367	19.167	12.860
14	Patapsco River	5.715	1.310	1.610	10.333	-	2.524
15	Big and Little Annamessex Rivers	23.030	18.837	19.705	21.923	18.463	11.075
16	Gunpowder and Bush River Headwaters	2.389	1.000	1.000	3.078	-	1.000
17	Pocomoke Sound, Maryland	22.727	19.480	18.505	-	15.350	10.477
18	Magothy River	6.633	1.000	7.600	14.200	-	7.183
19	Severn River	8.040	2.060	8.307	9.200	-	8.469
20	Patuxent River	8.930	4.585	10.532	11.466	10.416	11.878
21	Back, Middle and Gunpowder Rivers	3.288	1.000	1.000	5.159	3.814	1.000
22	Curtis and Cove Points	15.589	10.700	12.600	-	10.367	12.371
23	South, West and Rhode Rivers	9.212	5.940	9.110	-	9.337	9.950
24	Chester River	11.703	9.239	7.197	12.182	7.419	4.851
25	Love and Kent Points	17.225	14.375	9.762	15.587	9.400	6.375
26	Smith Island. Maryland	23.641	18.691	21.492	22.618	12 700	13.953
	AVERAGE	15.441	9.663	10.371	13.490	10.956	8.487

[a] Migratory Bird and Habitat Research Laboratory files 1977

Table 82. Average monthly[a] salinities (ppt), Chesapeake Biological Laboratory, Solomons, Maryland, 1970-1976[b]

Year	Salinity (ppt)												
	Jan	Feb	Mar	Apr	May	Jun	Jly	Aug	Sep	Oct	Nov	Dec	Average
1970	16.0	13.9	12.5	7.5	-	11.8	11.9	13.5	14.9	16.0	15.7	15.5	13.56
1971	13.5	15.0	10.3	9.3	11.5	10.3	12.8	13.3	14.4	-	-	13.9	12.43
1972	12.5	12.0	11.0	9.0	8.8	9.5	3.5	7.0	10.3	13.1	14.5	9.4	10.05
1973	9.0	8.8	10.0	10.5	10.8	8.5	10.0	10.6	12.5	14.0	14.5	15.0	11.18
1974	10.5	11.0	11.5	9.5	9.1	9.5	11.0	12.9	13.5	14.9	18.9	18.0	12.53
1975	16.5	-	10.5	9.0	8.5	10.5	9.8	9.8	13.1	9.6	10.5	9.9	9.81
1976	11.0	8.4	7.0	8.3	8.6	9.2	9.9	12.0	12.8	13.0	11.5	11.5	10.27

[a] Salinity measurements taken on the 15th day of each month

[b] Chesapeake Biological Laboratory files 1977

FAUNA

Cyprinus carpio, or European carp, is known for its destructiveness to aquatic vegetation in the Chesapeake Bay. Their feeding patterns in the soft muds for mollusks and crustaceans can cause turbidity severe enough to eliminate aquatic vegetation (Uhler 1958). The physical uprooting of the plants depletes the beds, but according to Martin and Uhler (1939), the exclusion of light due to the "roiled" waters is the major factor in plant mortality. Uprooted plants then float to the surface shading out remaining plants and increasing the turbidity through detrital output (Sills 1970). Uhler also reported that in the late 1950s submerged vegetation was almost wiped out by carp on the Susquehanna Flats. In an area of Nomini Creek in the lower middle Potomac River, a mat of *Myriophyllum* was reported to have been cleaned out by carp in May of 1968 (Steenis et al. 1972). However, the activity of *Cyprinus carpio* may have been a contributing factor in the dispersal of milfoil from 1958 to 1965 in the Chesapeake Bay (Bayley et al. in press).

The cownose ray (*Rhinoptera bonasus*), a summer inhabitant of the Chesapeake Bay, has been linked to the disappearance of *Zostera* beds. The decline of the eelgrass in late August and early September of 1973 coincided with sightings of large schools of rays (Orth 1976). In search of hardshelled mollusks, primarily bivalves, which it crushed between its dental plates, the rays uncover their prey with vigorous digging by their pectoral fins (Orth 1975*a*).

Their actions were so extensive that large patches of the Zostera beds were completely uprooted, leaving no roots or rhizomes for regrowth. Depressions in the "pock-marked" bottom were generally a meter wide and 20 to 45 cm deep. Core samples revealing a mixture of sediment and buried decaying Zostera leaves and roots provided direct evidence of the cownose ray's disturbance. Upheaval of the Zostera communities greatly reduced the species diversity and density of the infaunal inhabitants indigenous to these beds (Orth 1976). Orth (1975b) believes that the blue crab, Callinectes sapidus, may have been another agent of import in Zostera declines due to an enhanced ability to dig for food in sparsely vegetated areas.

Another potentially destructive group of organisms to the aquatic plants are the crustaceans, specifically crayfish. Several species are known to feed extensively on the submerged vegetation and have been incorporated into weed control programs (Martin and Uhler 1939). Although most crayfish are omnivorous scavengers, they revert to vegetarianism when necessary. Orconectes causeye, a native to the Western United States and a natural herbivore, was found most effective against Potamogeton sp. (Commission on International Relations 1976).

In addition to aquatic organisms, terrestrial mammals are consumers of submerged vegetation. Muskrats may devour emerging plants. Deer have been known to feed extensively on aquatic plants when deer populations are especially abundant. Potamogeton spp. are usually abundant in beaver flowages, especially the older ones (Linduska 1964).

Various species of waterfowl are well known for their food preference for for SAV (see Chapter 2). Waterfowl populations have been seen to cycle with SAV populations fluctuations. This relationship between SAV and waterfowl has been of long standing and there is no indication that waterfowl have been overrazing the Chesapeake Bay in recent years.

In conclusion, however, most documented destruction of SAV by fauna is for comparatively small areas. It is implausible at this time to attribute the massive decline in Bay rooted aquatics to grazing activities of any animals.

SEDIMENTS

The importance of sediments, their physical characteristics and chemical composition, in influencing the distribution of submerged aquatic plants has long been recognized. Early workers (Pond 1905; Pearsall 1920; Misra 1938) in their analyses of factors that determine the distribution of freshwater macrophytes, were of the opinion that the nature of the substrata was the single most important variable. This rather simplistic conclusion failed to consider the interactions between sediments and other chemical, physical and biological forces.

The role of sediments in supporting submerged rooted plants is two-fold. The substrata serves a physical function as a medium for mechanical support of the plant. This was at one time thought to be the primary function, and that the roots of submerged plants, with a lack of an advanced vascular system, only anchored the plants (Welch 1935). However, the work of Pond (1905), Pearsall (1920) and others suggested that rooted macrophytes derived some of their nutrition from the sediments. Recent work with radioactive tracers has confirmed this (McRoy and Goering 1974; Demarte and Hartman 1974). Thus, the second role of sediments is in the mineral nutrition of the plant species. The distribution of submerged aquatic vegetation is dependent on the ability of the sediments to not only provide mechanical support but also nutrients.

The sediment itself is derived from eroded bedrock, terrigenous input and biological activities within the aqueous environment. This material is sorted by turbulent activities with coarse sediments being deposited first and followed by progressively finer sediments. In lentic systems, this results in a gradation of coarse sediments inshore and finer particles farther out. In a tidal estuary, finer particles are desposited in the upper reaches of high tide according to the scrub and scour lag theory of Postma (1967). These areas thus serve as sediments "traps".

Besides providing firmer root support, finer sediments possess a greater surface area for adsorption of various compounds and allow a high cation exchange capacity. Nutrients such as ammonium ions and various phosphate compounds may enter the aqueous system adsorbed onto sediment particles and become incorporated into the sediment proper where they can be available to rooted aquatics. Many pesticides and herbicides also bind readily to fine particles and may reach greater concentrations in sediments composed of these particles.

Organic matter introduced into the sediments creates an oxygen demand through its decay. This leads to the formation of anaerobic sediments in submersed soils where oxygen input is limtied. Anaerobic soils enhance the availability of certain nutrients. However, too much organic matter such as found in peat formations possibly could deplete oxygen to a point where existence of submerged aquatics in unlikely. Loss of nitrogen through denitrification may also be significant.

Submerged aquatic vegetation is generally absent in areas of high turbulence (Sculthorpe 1967). In the Back Bay area, soils which are potentially the most productive are in actuality the least productive due to turbulence which tends to resuspend fine sediments. Allocthonous siltation may modify the sediments to a point where a change in vegetation patterns becomes evident. Jones (1949) attributed the absence of submerged vegetation in a Welsh river to an unstable, shifting sediment caused by siltation from mining wastes. Heavy storms may alter sediment patterns by input from the land and may have more impact than "normal" runoff. For example, Hurricane Agnes dropped more suspended matter into the upper Chesapeake Bay in a 10-day period than in the previous 10 years (Schubel 1974). However, submerged plant communities tend to self-maintain in regards to shifting sediments by the stabilizing action of roots, stems and leaves. A change in bottom sediments from soft to hard may also affect vegetation patterns (Cronin 1976).

In general, submerged aquatic plants are unable to colonize such coarse substrates of boulders, stones and gravel and prefer stablized fine sediments such as would make up muds. Under both field and laboratory conditions, Misra (1938) found the greatest growth of Potamogeton perfoliatus to occur in sediments which he termed "moderately organic black flocculent mud" with a humus content of 12.26 percent. Elodea canadensis was found in sediments of the same physical type but with slightly higher organic content (20 to 40 percent) in the same study. Martin and Uhler (1939) found that while firm sand would support the growth of Potamogeton, Vallisneria and Ruppia maritima, this was not the optimum substrate. In addition they showed a retardation of growth in sediments of high organic content due to the exclusion of light by dissolved matter.

In the Back Bay area of North Carolina, clay and shell substrates were found to be poorly colonized by submerged aquatics as were peat sediments (Bureau of Sport Fisheries and Wildlife et al. 1966). Silt and loam were the preferred media followed by sand. In this study, Vallisneria and P. pectinatus were found to occur on sand, silt and loam at all depths. P. perfoliatus and Najas were more abundant on silt and loam than on sand. R. maritima and Chara occurred most frequently on sand, followed by shell, clay and loam and were rarely found on peat, muck or silt (Bureau of Sport Fisheries and Wildlife et al. 1966).

Fenwick (unpublished) in a survey of submerged aquatic vegetation of the Chesapeake Bay found that finely divided mud supported the best growth. Of stations examined which had at least partial mud substrate, 67.7 percent possessed vegetation. Sand was found to be the next best (61.8 percent were vegetated) and peat sediments were colonized the least (12.9 percent).

The sediments thus appear to be instrumental in the distribution of submerged aquatic vegetation. However, the compostion and type of sediment is itself controlled by a multiplicity of interactions of a physical and chemical nature, many of which also affect the submerged aquatics directly. In summary, the effect of the sediments on Chesapeake Bay submerged aquatic disappearances is probably not a direct one, but may be extremely important in the adsorption and transport of other toxic substances, in SAV occurrence and in light exclusion discussed previously.

TEMPERATURE

The extent to which temperature in the aquatic environment influences the distribution of hydrophytes is limited because of less severe temperature fluctuation compared to an aerial habitat. Water serves as a buffer for temperature. Diurnal fluctuations, however, may have a significant impact on the metabolic processes due to the interactions of temperature with the dissolved oxygen concentration and nutrient supply (Sculthorpe 1967) and carbon availability as well. The use of water for irrigation practices, dams and industrial wastes can result in thermal loadings to the Bay. Mihursky (1969a, p. 6) defines thermal loading as"... a man-imposed, excess rise in waterway temperature which may gravely threaten aquatic life involved". Steam electric stations are considered to be especially harmful since an average 335 megawatt (MW) conventional coal-fired generating station requires under winter operation 950,000 liters of water per minute with temperature increase of 12.8 C between intake and effluent in a normal estuarine one-pass design. During other seasons the steam electric station (S.E.S.) industry pumps through 1,900,000 liters of water per minute, heating the water to an average of 6.4 C above ambient (Mihursky 1969b). For every kilowatt (KW) of electricity produced, two KW of energy are discharged as waste heat (Mihursky et al. 1970). The discharge of this effluent forms a plume-like pattern of warm water over the source (Mihursky 1969b) which eventually becomes confined to the shorelines (Anderson 1966). An atomic or nuclear S.E.S. is about 32 percent efficient in energy conversion compared to an approximate 40 percent efficiency of a fossil fuel S.E.S. (Mihursky and Cronin 1974). Atomic plants require more cooling water per MW than fossil fuel plants (Parker 1965) and produce approximately 50 percent more waste heat per unit of electricity (Mihursky 1967). For the nuclear plants to economically compete, they must be of large capacity of the order of 1,000 to 4,000 MW requiring up to 265 billion liters of water per day for cooling (Mihursky 1967). With electrical requirements ever increasing, Mihursky (1967) projected that this demand will double every six to ten years.

Other than the dissolved oxygen concentration, temperature governs numerous interdependent factors such as concentrations of carbon dioxide, pH, toxicity and biochemical reactions (Hoak 1961). Any heating four feet below the surface can be considered the beginning of a serious situation (Anderson 1966). The equilibrium concentration of dissolved oxygen is inversely proportional to the temperature (Hoak 1961). When passing through a condenser in a power plant, the oxygen content of the water is reduced. This reduction could be serious, but studies reveal that this lost oxygen is quickly replenished (Engle 1961). The distribution of effluent is characteristically patchy in estuarine systems due to the complexities of flushing and layering which may be encountered, sometimes causing locally lethal temperature conditions (Naylor 1965). Lethal levels, either high or low, can directly cause mortalities, influence daily and seasonal behavior or influence distribution and abundance in plant popluations (Mihursky and Pearce 1969 ; Mihursky 1967). Although acclimatization is suspected (Naylor 1965), unsuitable temperatures are generally tolerated rather than compensated for which could affect metabolism, reproduction, etc. (Anderson 1966). Setchell (1924) found that above 25 C, anthesis of *Ruppia maritima* was slow. If the temperature continued, the process ceased altogether. However, in temperatures from 22 to 24 C, *Ruppia* flourished and bore fruit (Setchell 1924). Ritchie and Genys (1975) report temperatures as high as 30 C in the Patuxent River where Anderson (1966) investigated the disappearance of *Ruppia maritima* across the river from the effluent canal of the Chalk Point generation plant. Anderson (1966) found that *Ruppia* is sensitive to sudden changes in its environment especially during periods of seed germination and the production of new shoots from rhizomes. Recorded temperatures of 25 to 26 C in May of 1965, which are borderline for new shoot production in *Ruppia*, are apparently significant enough to cause this plant's disappearance. Although other contributing factors are possible for *Ruppia's* decline, considering its sensitive nature, Anderson (1966) indicated that the raised ambient temperatures from power effluent input was the primary cause of decline. Potamogeton perfoliatus increased in coverage after opening the Chalk Point power plant indicating the species tolerance for elevated temperature and salinity (Milhursky 1969b). Anderson (1969) concludes that P. perfoliatus became physiololgically adjusted to atypically high environmental temperatures. Ceratophyllum demersum and water plants in general are more easily damaged by environmental changes than terrestrial plants.

This substitution of one species for another is an example of factor compensation (Odum 1971). However, the massive SAV decline of the 1970s shows no indication of larger scale species substitution--almost every species is negatively affected. This seems to indicate a different type of a problem than temperature. Furthermore, although Orth (see Chapter 1) believes that the increasing winter water temperatures in the early to mid 1970s may have caused Zostera declines in Virginia, the temperature data (Figure 16) for Baltimore and Solomons indicate much less of a rise in Maryland waters. In fact, when the MBHRL occurrence data is plotted against yearly temperature averages at Solomon's Maryland, a positive low correlation is obtained (r=.16). This indicates that temperature rises in the upper Bay seem to be accompanied by slight rises in SAV. However, the relationship is so weak that it is probably insignificant biologically. When the monthly temperatures at Solomons during the growing season were plotted against percent occurrences of SAV, the following "r" values were obtained: April, +.53; May, +.48; June, +.51; July, +.12; August, -.44; and September, +.94. Although April through July and September show a positive correlation similar to the yearly upper Bay temperature trend, the August temperatures are inversely correlated. This finding is interesting since it substantiates field observations that August temperatures seem to regulate SAV population declines. However, the magnitude of the correlation is not high enough to attribute the 1970s decline to summer temperatures.

DISEASES

The disappearance of the submerged aquatic vegetation of the Chesapeake Bay has been attributed to several physical and biological factors. In a few cases specific organisms have been found to be associated with the population decline of certain plant species within the Chesapeake Bay.

Rhizoctonia solani, a fungus found to be pathogenic to Potamogeton pectinatus, is generally found worldwide in areas where the staple crop is potatoes. Bourn and Jenkins (1928) concluded that Rhizoctonia was the most important factor in the disappearance of aquatic duck food, though Lumsden et al. (1963) questioned the extent of its importance. Bourn and Jenkins (1928) determined that this pathogen would kill aquatic plants and attack such terrestrial plants as the potato. An additional strain of R. solani from diseased potato plants was found to attack aquatic plants. Penetration of R. solani into its hosts by hyphae is commonly through the intact epidermis from beneath dome shaped infection cushions, or directly through openings and wounds (Joyner and Freeman 1973). Experimental evidence showed that in June diseased plants were coated with gelatinous remains of old hydroid colonies. Subsequently, dark brown lesions appeared on the lower stems. From August to September the effects were more acute and resulted in the complete browning and death of entire aquatic plant beds. Although Bourn and Jenkins (1928) found that increased salinity between 2 and 7 ppt lowered plant resistance, Lumsden et al. (1963) found salinity inhibitory to R. solani at 4 to 6 ppt. Studies with water hyacinth revealed increased susceptibility at 28 C which decreased at 30 C or above (Joyner and Freeman 1973).

Rhizoctonia solani has been found to attack the majority of the species of duck food plants with Potamogeton pectinatus being the most susceptible. Vallisneria and Najas are not readily infected unless associated with Potamogeton. Ruppia maritima was found to be the most susceptible next to Potamogeton but usually survived infestation due to its reproductive powers. Susceptibility generally decreased when plants grew in sandy soil somewhat sheltered from the winds (Bourn and Jenkins 1928).

The 1961 blight of Potamogeton pectinatus in North Carolina was found to result less from the presence of R. solani and more from combined environmental factors along with one or more Phythium species. Three unidentified Phythium species were determined with Phythium-3 being the most likely infectious agent. Areas of higher salinity resulted in improved plant condition, however this increased resistance to disease infection could have been the result of more favorable environmental conditions for the plants (Lumsden et al. 1963).

Prior to 1963, Myriophyllum spicatum inhabitated over 40,000 ha in the Chesapeake Bay (Elser 1969). Its subsequent decline has been attributed to Lake Venice Disease and/or Northeast Disease whose responsible pathogens have yet to be isolated. Elser's study revealed that by 1967 healthy milfoil was nowhere to be found in Maryland with most beds showing symptoms of both diseases.

Lake Venice Disease manifests itself as a brownish, silt-like coating on leaves and stems that causes a gradual wasting away of the host. If flowering occurs, it is very sparse (Elser 1969). Only under low light intensities (indirect light) in the labortory well plants produce symptoms resembling Lake Venice Disease, and the disease cannot be mechanically transmitted (Bean et al. 1973). Bean et al. concluded that only after low light intensity exposure does Myriophyllum become susceptible to Lake Venice Disease. The casual agent may be a bacterium, virus or fungi.

The Northeast Disease which attacks milfoil was extensively researched by Bayley et al. (1968) who concluded that the disease was a virus, virus-like particle or a toxin. Dr. K. Corbett (Virologist, University of Maryland) has suggested that a causal agent may not exist, but rather the disease is a result of a combination of environmental conditions (personal communication). The Northeast Disease symptoms are: initially broken leaflets; stems and leaves turn dark brownish green and become stiff; and the leaves gradually drop off leaving a blackened stem. Distortion of the petioles is commonly seen when the diesease progresses slowly (Elser 1969).

Prolific growths of *Zostera marina* were characteristic of the Atlantic Coast of the northern United States and Canada prior to 1931. At the same time the West Coast of the United States and European *Zostera* beds were also dimishing in size. Local declines were noticed as early as 1930. A devasting disease which has inconclusively been attributed to *Labyrinthula microcystis* almost completely destroyed the *Zostera* beds over much of the Atlantic Coast of the United States from Beaufort, North Carolina to Nova Scotia by 1932 (Tutin 1934; Mackin unpublished). Destruction of the eelgrass beds resulted in the washing away and erosion of previously bound sediments changing their configuration and productivity (Linduska 1964). The disease has been described as "epiphytotic" and "enphytotic" for various areas. In recent years, some recovery has occurred both in the United States and Europe (Mackin unpublished).

The infection is not usually visible on new tissue, however, older leaves display dark splotching and/or black lower leaves. Although usually confined to the leaf epidermis, spotting may occur in any green tissue. Streaking is the result of the infection of the larger, longitudinal mesophyll cells ending at the lateral septa. The stem cuticle, cortex and rhizomes may also become darkly discolored (Renn 1935; Tutin 1934).

According to Young (1943) the predominant morphological stage is the neoplasmodium. In this stage masses of filaments form an "intricate lacy network" on which tapering cell bodies migrate singly or in "rope-like aggregates". Sorus and vegetative cyst formation is a characteristic form in which the disease is believed to be transported by water currents (Mackin unpublished). The mechanism of motion is unknown, but the spindles produce filamentous pathways on which the organisms seem to glide. Apparently, nutrition is obtained through extracellular digestion and absorption in solution (Young 1943).

Labyrinthula has been found to be extremely tolerant of its environment. Activity occurs in a range from 0.3 to 27 C, but the optimum temperature appears to be between 14 and 24 C. This optimum correlates with the fructification of *Zostera*,blighting it just prior to propagation. The pH tolerance ranges from 4 to 9 without great alteration of the organism's activity. Roughly the salinity optimum is from just below seawater to quite a bit above it. The temperature and salinity preferences are similar to the conditions prevalent during the recorded periods of massive infestation during the summer months. Additional hosts which are generally closely associated with eelgrass beds include: two algae, *Cladophora hirta* and *Chaetomorpha* spp. *Linum*; and *Zannichellia palustris* var *major* and *Ruppia maritima* var. *rostrata* (Young 1943). Mackin (unpublished) also cites *Fucus furcatus*, *Ectocarpus conferoides*, *Cystocionium perpureum* (all algae) and *Zostera hornemanniana*. This wide host range could insure the continued presence of *Labyrinthula* (Young 1943).

Another suggested causative agent for the disappearance of *Zostera* is the fungus *Ophiobolus halimus*. Its abundance and activity is variable, especially in the United States, and is not necessarily found in areas diseased by *Labyrinthula* (Renn 1935). Mackin (unpublished) stated that its importance in the wasting disease of eelgrass in the United States was improbable since it was so rarely found. Renn (1934) restricted *Ophiobolus* effects to Canada and northern Europe.

A considerable amount of controversy exists over the causative agent(s) for the disappearance of Zostera marina. Labyrinthula has not been conslusively determined as the causative agent and Ophiobolus has been almost eliminated from consideration. The salinity and temperature fluctuations have also been considered for the decline. Butcher (1935, p. 545.) summed up the situation by concluding that the disappearance of Zostera was more than likely due to a "large number of circumstances and not...a single catastrophic event".

Elevated temperatures can result in a sub-optimal physiological condition enabling opportunistic pathogenic organisms to overcome their hosts. Since many pathogenic organisms are ubiquitous and constant components of the aquatic environment, the development of stress conditions, i.e. elevated temperatures, can result in infection perhaps leading to significant declines in areas affected by thermal plumes (Mihursky et al. 1970).

Field observations in the 1970s have not indicated any recurring disease-like symptoms on SAV. In the summer of 1977, Suzanne Bayley examined an extensive bed of Myriophyllum which looked brownish, but concluded it was not diseased. Rather the brown film on the leaves and stems was a mixture of periphyton, detritus and sediment. The involvement of SAV as a group in recent declines would seem to indicate that disease alone has not been responsible. The likelihood is small that all the species in the Bay could be hit by a single pathogenic species. It is possible that some major ecological change could have resulted in the lowering of plant resistance thus allowing invasion by a pathogen or series of pathogens. This environmental change could have been one or more of such factors as turbidity, chlorine or agro-chemicals.

LOCAL ECOLOGICAL FACTORS

There are several environmental factors that negatively impact submerged macrophytes on a localized level. Included among these are dredging and boat traffic. Direct action from these sources can cause uprooting and physical damage to grass beds. However, indirect impacts can be caused by the reintroduction of bottom sediments into the water column thus increasing turbidity. The degree of impact from suspended sediments is related to bottom sediment composition. Fine, silty organic materials require more time to settle out than heavier, sandy material.

Dredging

A hydraulic clam dredge, for example, uproots all vegetation in a 75 to 90 cm wide path. This apparently happended at the north side of Cambridge Bridge on the Choptank in the 1960s. In very shallow water, boat propellers also uproot SAV. The extent and permanency of damage to SAV beds depends on the reproductive means of the species involved. Species that normally reproduce only sexually could be virtually wiped out by extensive dredging. Species capable of vegetative reproduction have better chances for survival.

Manning (1965) discussed experiments conducted in the Patuxent River and the Eastern Bay in 1956 to determine revegetation requirements for areas stripped of vegetation by clam dredges. In comparing two sections of the Patuxent River, one area where thousands of bushels of clams had been dredged and another area where dredging was prohibited, the conclusion was reached that natural forces and conditions were responsible for SAV distribution and abundance rather than clam dredging. No long term impacts from dredging activities could be proved since the prohibited area showed less vegetation than the dredged one.

Dredging for the purpose of increasing ambient depths completely removes existing vegetation and alters the habitat. Macrophytes normally colonize the shallower areas along the shoreline and extend out into deeper water based on the photic zone. By increasing the depth, SAV would be prohibited from recolonizing due to a decrease in the amount of light reaching the new dredged bottom depth. However, dredging results in piece-meal destruction of a localized nature rather than in Bay-wide impacts.

Boat Traffic

Damage to submerged macrophytes from boat propellers is essentially of local importance. However, with the increased number of pleasure boats that presently navigate the shallow water areas of the Chesapeake Bay estuary, damage to existing beds could be considered to be more extensive.

Figure 15 (Chapter 1) shows an aerial photograph of an eelgrass bed in Mobjack Bay, Virginia, that has been impacted by boating activity. The crisscrossing pattern of light streaks indicates boat propeller damage. This destruction is usually seasonal but excessive propeller disturbance over an extended period of time would continue to limit SAV on a localized basis.

WATER MOVEMENT - TIDES AND CURRENTS

Submerged aquatic species do not normally colonize areas subject to continuous strong currents or tides. Excessive water movement tends to scour the bottom to such an extent that submerged macrophytes are prohibited from colonizing. Such water movement also resuspends fine sediment particles and contributes towards increased turbidity.

Daily tidal flushing is necessary within an estuary in order to remove metabolites and bring in nutrients. This was thought by Teal (1962) to be responsible for the high productivities in a tidal salt marsh and was found necessary for aquatic plant community development in the South River (Philipp and Brown 1965). Submerged plants are exposed to daily variations in light quality and quantity as a result of tidal currents. In extremes this can be detrimental (Joanen and Glasgow 1965). Extreme low tides cause plants to become exposed resulting in dessication. Extreme high tides raise ambient water depths and effectively decrease light penetration to the benthos.

However, a scan of tide records from the Chesapeake Biological Laboratory at Solomons, Maryland did not show any abnormal tide events which did not also occur in previous decades. Therefore, this can be safely ruled out as a major factor in the SAV declines of the 1970s.

Extreme water movement resulting from storm events can cause substantial physical damage to existing aquatic beds. Field observations by McCann (1945) determined that the branch tops at the surface are often fractured by wave action. In the case of Ruppia maritima, these detached fragments are not capable of survival. Such storm events as Agnes in 1972 and Eloise in 1975 may have negatively impacted Bay submerged vegetation; however, historic accounts of previous major storm events do not indicate vegetation losses from storm damage alone. This is probable because submerged aquatics can quickly colonize areas by vegetative reproduction (see Chapter 1). Thus physical disturbance is a stress factor which these plants are well adapted to.

pH

As discussed in Chapter 2, the photosynthesis and respiratory activity of submerged aquatic flora causes diurnal fluctuations in the dissolved CO_2 content of an aquatic environment. pH, in turn, affects plant enzyme activity, seed germination and variety of other plant responses.

The normal pH of the Chesapeake Bay is from about 6 to 9. Drastic fluctuations could cause damage to SAV; however, given an otherwise favorable environment and sufficient necessary nutrients, most SAV species should be able to tolerate a wide pH range. Extreme pH fluctuations can probably be found as a localized condition resulting from industrial effluent input. Such a situation would be of a point source nature and would affect native vegetation only in the immediate vicinity of the effluent sources. No documentation has been found as to whethkr pH fluctuations have affected SAV in the Chesapeake Bay.

CARBON DIOXIDE SOURCES

Submerged aquatic species have potentially three entry ports for the CO_2 necessary for photosynthesis. Aquatic plants are capable of utilizing dissolved CO_2 in the water column through leaf uptake. They are also able to take up bicarbonate ion (HCO_3^-) for usage, and finally CO_2 may possible be taken up from the sediments by the roots and transported to the leaves. These three possibilities are of interest because they furnish the most crucial raw material for photosynthesis in the stuarine environment--carbon. Any limitation in the carbon availability would be reflected in an intermediate decline of SAV productivity. The problem is that carbon dioxide dissolved in water has a much lower coefficient of molecular diffusion than under gaseous conditions. This situation is complicated by the existence of a boundary layer of unstirred water near the surfaces of SAV leaves (Gessner 1955, cited in Sculthorpe 1967). These physical conditions lead to a very slow entry rate of CO_2 into aquatic plants compared to land plants under the same concentrations of free CO_2.

This limited accessibility has given an evolutionary advantage to those species which maximize surface area for absorption of CO_2. Dissolved CO_2 appears to be the preferred carbon source for submerged aquatic vegetation (Steeman Nielson 1946).

It has long been speculated and argued that aquatic plants possess the ability to utilize bicarbonate ions for subsequent use in the Calvin cycle. A historical review on the subject is presented by Sculthorpe (1967). Hutchinson (1975), in a synthesis of the various experiments performed in this regard, concluded that of the flowering aquatic plants which occur in the Chesapeake Bay, the following can use bicarbonate as a source of carbon: Ceratophyllum demersum, Myriophyllum spicatum, Vallisneria spiralis, Elodea canadensis and Potamogeton pectinatus. The selective advantage that this ability confers is significant. When the pH rises above 9.0, free carbon dioxide does not exist, and plants existing at levels higher than this would have to be able to use HCO_3^-. Assimilation in more acid conditions could occur, even if the CO_2 levels were removed to a point of depletion. In most aqueous systems, the concentration of HCO_3^- is likely to be several times and in some cases several hundred times the concentrations of CO_2. The advantage of being able to draw on this reserve in periods of low dissolved CO_2 is obvious.

Bicarbonate ions are taken up through both sides of the leaf of Potamogeton as the calcium salt. This uptake is balanced by the release of $Ca(OH)_2$ through the upper surfaces, and the reaction is light dependent (Hutchinson 1975). The uptake mechanism and subsequent usage of bicarbonate can be thought of as a rough aquatic analog to the 4-carbon Hatch-Slack pathway evident in some terrestrial species. Carbon dioxide in both cases is complexed with a "carrier" compound and actively transported to a location where decarboxylation can occur and CO_2 entered into the Calvin cycle. The chemical nature of the carrier in aquatic species is unclear at this time.

Wium-Anderson (1971) working with an aquatic genus not covered by this volume, Lobelia, has determined that root uptake of CO_2 occurs followed by translocation to the photosynthetic site. That this occurs in other macrophytes has not been determined, however, a suggestion of this is seen by Hutchinson (1975) in the presence of many chloroplasts in the cells surrounding air spaces in the submerged stems of Myriophyllum. This pathway may be significant in environmental situations where CO_2 levels in the water column are low and those in the sediments high.

Submerged aquatic vegetation may be carbon limited under certain environmental conditions. There is a close association in natural sea water between salinity, pH and bicarbonate supply. A drop in salinity results in a pH drop and a shift in carbon form away from bicarbonates and thus may limit photosynthesis (Hammer 1968; Ogata and Matsui 1971). The shift in carbon source is likely to be more of a limiting factor than the osmotic pressure differential which also accompanies lowered salinity. Hammer (1968) maintained photosynthetic levels while dropping salinity by the addition of bicarbonates.

Changes in submerged vegetation patterns noticed in recent years may hinge on carbon availability. Salinity in the Chesapeake Bay has decreased somewhat recently from an average of 15.441 ppt in 1971 to 8.487 ppt in 1976. Following the argument presented above, the carbon balance has then been shifted in favor of CO_2. Orth (unpublished) believes temperature to be an important factor in the decline of SAV. The rise in temperature he has noted would decrease the solubility of CO_2 thus making it less accessible to macrophyte species. Nutrient inputs could ehnance algal species that, because of their favorable surface to volume ratio, could compete effectively with aquatic macrophytes for available CO_2. More research involving carbonate levels needs to be done to determine the relative importance of this factor in assessing SAV declines.

HEAVY METALS

Heavy metals occur naturally in marine, brackish and fresh waters generally in increasing concentrations with decreasing salinity (Bureau of Land Management 1976). Table 83 lists various common heavy metals with estimated values of natural occurrence in marine and fresh waters. Higher values in estuarine waters result from man-oriented sources as point and nonpoint pollution such as municipal wastes, industrial effluents, agricultural drainage and petrochemical products (Schroeder 1977; Sculthorpe 1967; Bureau of Land Management 1976).

Heavy metals generally are resistant to biological or chemical degradation (Bureau of Land Management 1976). Once introduced into an aquatic medium, soluble heavy metals tend to become adsorbed, complexed or precipitated onto particulates. Adsorption may occur not only to sediment particles but also to phytoplankton (Schroeder 1977). Not all metals are adsorbed equally nor are heavy metals associated solely with particulates. Small amounts of heavy metals can remain in solution (Bureau of Land Management 1976).

The role of heavy metals in plant metabolism is not fully understood. Some relationships, however, have been established. For instance, iron appears to be necessary for chlorophyll synthesis; manganese is essential for respiration and nitrogen metabolism; copper is an important enzyme component; and zinc is involved in auxin synthesis (Devlin 1975).

The normal uptake of heavy metals by submerged aquatic vegetation frequently results in element concentration in plant tissue. Hutchinson (1975) cites the work of numerous scientists substantiating the concentration of vanadium, copper and zinc by _P. pectinatus_, copper and zinc by _Vallisneria_ sp. and copper by _Chara_ sp. Both _Elodea_ and _Myriophyllum spicatum_ were found to concentrate iron. _Ceratophyllum demersum_ has been tested extensively for metal concentrations and found to accumulate copper, zinc and chromium. Concentration factor ratios of element concentration in organism to element concentration in environment have been documented in _Ceratophyllum demersum_ for arsenic (1,100 to 1,700 ppm), cadmium (500 to 700 ppm) and mercury (about 70 ppm). Experiments with _Zostera marina_ in Alaska (Bureau of Land Management 1976), have shown that zinc, copper and cadmium can become concentrated in roots, rhizomes and leaves.

It has been previously established that SAV plays an essential role in the food web through consumption of live material and detritus by herbivores and filter feeders (see Chapter 2). Due to the ability to concentrate heavy metals, SAV can serve as a vehicle for the recycling of heavy metals to higher trophic levels. This can result in increased and possibly toxic levels of heavy metals

Table 83. Naturally occurring soluble concentrations of various heavy metals in seawater and United States rivers

Element	Concentrations in seawater ppb[a]	Concentration ranges in U. S. rivers ppb[b]
Mercury	0.03	<0.1 to 17
Cadmium	0.1	<0.1 to 80
Zinc	4.9	1 to 800
Copper	0.5	0.9 to 12
Nickel	1.7	0.3[c]
Lead	0.03	3[c]
Silver	0.04	0.1 to 0.55
Cobalt	0.05	0.037 to 0.44
Iron	2	< 1[c]
Manganese	0.2	<1 to 180
Chronium	0.3	0.1 to 2.46

[a] Brewer 1975 (cited in Schroeder 1977)
[b] Kopp and Droner 1968; Turekian 1971 (cited in Schroeder 1977)
[c] Typical concentration

in aquatic fauna and man. However, without SAV to aid in recylcling metals, bottom sediments in the vicinity of heavy metals input can become a metal sink of sufficient toxicity to be unable to support any life (Davey and Phelps 1975).

Toxicity of heavy metals to species of submerged aquatics has not been established. Hydroponic studies by Lee et al. (1976) determined that several species of marsh plants were sensitive to heavy metals. They found that levels of iron and phosphorus in plant roots were related to the accumulation of zinc, nickel, lead and chromium. The absorption and translocation of phosphorus may be influenced by heavy metals. Sensitivity to heavy metals may also be greater in plants with a lower capacity to absorb and translocate iron. Lee et al. concluded that phosphorus and iron levels in roots may serve as a mechanism to control heavy metal levels.

Based on the lack of knowledge of the biological impacts of excessive levels of heavy metals to SAV and the extent of heavy metal pollution in the Chesapeake Bay, it is not possible to correlate recent SAV declines to metal inputs. It is possible that Bay grasses may be accumulating heavy metals in toxic or near toxic concentrations; however, there is no data available to support or refute such a hypothesis.

PETROCHEMICALS

Increasing inputs of petrochemicals and petrochemical products into the nation's waters have become a cause for concern. Petrochemicals enter the aquatic environment from tankers, refineries, municipal and industrial effluents, boats, and urban and river runoff. Despite the extravagance of petroleum inputs from oil spills, an estimated 41.7 percent of the petroleum products that reach marine environments come from urban and river runoff. Farrington (1975) refers to this input as "chronic dribbling."

Rough estimates of annual petroleum inputs into the Chesapeake Bay (U.S. Army Corps of Engineers 1977) indicate approximately 47 percent comes from urban and river sources. Municipal and industrial sources account for about 44 percent. These percentages are based on a total input of about 37,200 metric tons per year. The remaining petrochemical inputs consisted of 0.8 percent from oil spills, 1 percent from ship generated wastes and 8 percent from boats.

Sediment sampling by the Annapolis Field Office of the U. S. Environmental Protection Agency (U.S. Army Corps of Engineers 1977) found a range from 420 to 81,220 mg/kg oil and grease in Baltimore Harbor sediments. "Oil and grease", as defined by EPA's laboratory analysis, included naturally occurring lipids and hydrocarbons. Thus in order to differentiate between naturally occurring and man induced oil and grease, EPA suggested that any oil and grease concentrations in sediments greater than 1,000 to 1,500 mg/kg would probably have resulted from petroleum contamination.

A study by the Chesapeake Bay Foundation (1977) concluded that from 1973 through 1976, 379,702 gallons of petroleum products entered the Bay from spills in Maryland during transport and handling. Virginia contributed 497,391 gallons from 1974 to 1976. Spills in Baltimore Harbor resulted in 40 percent of the Maryland input and 50 percent of the Virginia contribution occurred in the Elizabeth River.

Included in petroleum products are a number of different products: kerosene, gasoline, crude oil and fuel oil contain "lower molecular weight components" that evaporate quickly and are more soluble. Lubricating oils contain heavier molecular weight components that are not readily evaporated. Upon mixing with water and sediments, oil can be adsorbed directly onto sediments or fractionated with some components subsequently adhereing to sediments. Sediments are then available for resuspension and transportation (Farrington 1975). Crude oil contains thousands of compounds (Brown 1975). Some components can be biodegraded by such organisms as bacteria of yeasts. However, the impacts on marine organisms resulting from such microbial degradation are as yet unknown (Farrington 1975).

The impact of oil contaminated sediments on submerged aquatic vegetation appears to be largely unknown. Farrington (1975) suggested that marsh grasses may be made unsuitable as a habitat. This could apply also to submerged grasses. A further mode of impact to SAV could occur through the capacity of oil and oil contaminated sediments to concentrate heavy metals and pesticides. Brown (1975) cited Walker and Colwell (1974) and Seba and Corcoran (1969) as having found an association between increased oil polluted sediments and increased sediment concentrations of zinc, chromium, lead, copper, nickel, cadmium and mercury, aldrin, dieldrin, DDT and possibly lindane, chlordane and heptachlor expoxide.

Free oil may also coat vegetation and could impact SAV by physically blocking nutrient assimilation and gas exchange in above bottom plant parts. Cowell (1969) cited in Ecological Analysists, Inc. (1976) maintained that oil can be directly toxic and even lethal to salt marsh flora. This may also be true for submerged vegetation. Ecological Analysts, Inc. (1976) suggested that negative impacts to annuals would probably be greater than to perennials due to the capability of regrowth from rhizomes in perennial species.

Based on the lack of knowledge of the biological impact of petrochemicals and petrochemical products in an estuarine environment and the extent of oil polluted sediments in the Chesapeake Bay, it is not possible to correlate recent SAV declines with oil inputs. Extensive laboratory and field anlayses would be required before such a correlation could be attempted.

SUMMARY

The previous chapter provided evidence to support the conclusion that submerged aquatic vegetation in the Chesapeake Bay has declined over the past 40 years. Based on this conclusion, various environmental factors have been investigated in order to ascertain possible causes for the Bay grass losses. Factors that can negatively impact the aquatic environment are summarized on two levels--localized impacts and Baywide impacts.

Localized impacts generally include turbidity, chlorine, nutrient loading, boat activity and dredging. Turbidity, chlorine and nutrient loading, while affecting more than one sub-estuary, have not been found applicable to the entire Bay. Based on data from the late 1960s through the 1970s, a few areas in the Bay have shown increases in turbidity, yet most have decreased. Chlorine inputs into the Bay have risen along with increasing population in the Bay area. There seems to be some evidence that excessive turbidities and chlorine plus its by-products negatively impact SAV. However, experimental work is needed to determine the extent of the effects. There is little evidence at this point that chlorine has more than a localized impact, but research is presently in progress.

Excessive nutrient loading often results in plankton and algal blooms such as have been noted in the Potomac River. There is no evidence that such a situation extends beyond parts of sub-estuaries. The main stem of the Bay has high flushing rates, large size and good vertical mixing which give it a relatively high resiliency to sewage loadings from metropolitan point sources. Power boats can cause localized impacts to SAV beds through propeller damage. Dredging, whether for clams or for increasing water depths, effectively removes SAV from the dredge line.

Impacts applicable on a Baywide scope generally include salinity, temperature, HCO_3^-, fauna, epiphytes, disease, agro-chemicals and large scale weather events. The bicarbonate ion as a carbon source may be a factor in plant declines in the lower saline areas of the Bay, but probably is not relevant in the lower Bay where higher salinities are found. In these higher salinity areas, HCO_3^- is more readily available. Unfortunately little is presently known concerning the importance of the bicarbonate ion to SAV.

Salinity has been implicated in the SAV problem. Over the past six years, salinities in the Bay have decreased subsequent to a salinity increase observed during the 1960s. Such a decrease as is presently being experienced might have a slight impact on a species such as *Zostera marina* which thrives in high salinities. Most other species actually prefer fresh water and brackish situations and can easily tolerate salinity decreases.

Baywide impacts due to temperature are not supported by substantial evidence. There are no indications that ambient temperatures have risen sufficiently to impact Bay grasses except in thermal plumes and possibly in the lower Bay. There is also no evidence as to Baywide grass losses due to excessive faunal activity or epiphytic associations.

Investigations into the disease theory for SAV declines have determined that specific species are susceptitble to several pathogenic organisms. However, during the years when *Zostera marina* was declining and *Myriophyllum spicatum* was dying back, there was no indication that the involved "diseases" were pathogenic to the many other SAV species native to the Chesapeake Bay.

The impact of agro-chemicals on SAV is presently under scrutiny. Herbicide usage has increased heavily in the Bay area, yet an understanding of the fate and mode of action of herbicides in an aquatic environment plus their subsequent impact on SAV is not known. Herbicides, both dissolved and adsorbed to sediment particles, have been determined to negatively impact at least one species of SAV in bioassay experiments. But the nature and extent of such impacts in field conditions are not as yet known.

The force with which Hurricane Agnes, followed by Hurricane Eloise, hit the Bay area in 1972 has led some researchers to support the natural disaster theory. After past hurricances and droughts, there have been reports of low SAV populations, but recovery occurred in no more than one or two years. After Agnes, the low point in vegetation occurrence was in 1975 and any major recovery did not begin until 1976. Although some may argue a possible lag effect from Agnes, this does not seem likely in view of possible recovery rates if the submerged aquatic vegetation is otherwise healthy. Most researchers we have interviewed corroborate this view that SAV has changed dramatically, and there is probably some man-induced cause or causes.

Two additional factors that may cause negative impacts on SAV are heavy metals and petroleum products. There is little data available at the present time concerning the toxic effects of heavy metals or petroleum on SAV. Also, field sampling data is insufficient to determine whether or not toxic levels are present in the Chesapeake Bay.

The Pensacola Bay system in Florida has been experiencing seagrass losses since 1949. The areas where the grasses appeared to be the most heavily impacted were areas that had been the most altered by man and seemed to have the most beach erosion. Vegetation losses were attributed to the synergistic effects of sewage and industrial waste effluents, dredging and filling, beachfront alterations and changing watershed characteristics. It is possible that similar synergisms are responsible for the Baywide submerged aquatic declines.

CHAPTER 5

MODELS AS TOOLS FOR SYNTHESIS AND INTEGRATION: SUBMERGED AQUATIC VEGETATION IN CHESAPEAKE BAY

DEFINITION OF MODELS AND JUSTIFICATION OF THEIR USE

A model is an analogy which serves as partial representation of a real system (O'Neill 1975). However, it is not meant to be a miniature, one-to-one mapping of the real world, but rather a simplified abstraction (Patten, 1971). Scientists who deal with environmental systems may have elaborate mental models to describe their understanding of the complex systems that they research. There is, however, a limit to the capacity of the human mind. Moreover, if large-scale management decisions (such as those related to SAV) are to be based in part on scientific understanding of these complex environmental systems, it is important that the various pieces of information on those systems which are linked into each mental model, be pooled together and subjected to critical examination. As long as a model resides only in a scientist's intuition, it is not available for critique and comparison with other models. Models can serve as important tools in understanding a large body of scientific data by synthesizing information into a generalized framework. They provide a formal format for describing the structural and functional properties of systems and for summarizing data gathered in the study of such systems. Modeling a complex system such as the Chesapeake Bay in detail would in itself require a large scale effort, which is beyond the scope of this report. However, we provide here some demonstrative procedures and preliminary efforts directed toward understanding the Bay ecosystems and the inter-relationships of submerged aquatic vegetation.

GENERAL PURPOSES AND STRATEGIES FOR MODELING

From a theoretical stance, the general purposes of developing scientific (Levins 1966) models are to:1) explain how nature works; 2) predict effects of perturbations on natural system; and, 3) aid in management of natural resources. To meet these broad goals, models are designed for: realism (mechanistic consistency with scientific observations); generality (flexible consistency with a large spectrum of natural conditions or systems); and precision (ability to closely reproduce or mimic the behavior of natural systems). No single model can meet all three criteria with equal strength, so it is usually necessary to develop a group of models addressing the same resource management problem or scientific hypothesis from different points of view. The specific strategies of modeling are outlined below.

Obtaining a Systems View Within an Ecosystem

Ecosystem components such as SAV are very seldom affected in isolation of each other; a change in one usually affects others to varying degrees, often in unexpected ways. Because of these non-linear and feedback properties of environmental systems it is important to obtain a systems overview when investigating the affect of alterations to the environment. The use of model diagrams helps to maintain the emphasis on the overall and important aspects of the system, as well as on important components.

Promoting Understanding of the System Being Studied

The act of committing mental conceptions to paper forces the investigator to examine all relationships within the system. Difficulty in drawing the diagram points out an imcomplete understanding of the system that may not have been anticipated. Thus, the diagraming process is a tool for promoting, sharpening and clarifying relationships within a system.

Defining the System Affected

From ignorance, important effects of an environmental alteration are often not considered because they occur in some part of the system with which the investigator is not familiar. By preparing a diagram, connections to other parts of a system may become evident that would have been otherwise overlooked.

Organizing Knowledge

Evaluating environmental impacts usually involves data from a variety of sources. Use of a model can be a powerful tool for organizing this knowledge. In this way synthesis, which should be the cornerstone of environmental impact analysis, is facilitated because the system as a whole is more easily comprehended.

Plan Research Programs

Placing existing data onto a diagram during the early stages of an environmental analysis can provide valuable help in planning an effective field research program. The diagram will show the overall kinds of measurement that must be made and indicate data gaps, thereby helping to plan the most effective sampling program. Simulation of the model, which represents the initial understanding of how the system works, allows the investigators to test this understanding. If model output does not match the data being gathered, then it is likely that our understanding of the system is still incomplete. Research may then be redirected as needed. Location of sensitive parameters within the model help to concentrate the research effort on critical pathways.

Vehicle for Discussion

A diagram provides a common ground for discussion among all parties that is independent of specialized language because the model is an explicit expression of the system structure believed to exist.

Predict Impacts

A simulation model allows the testing of predicted effects. If behavior of the model validates fairly well against presently observed conditions in the environment, then some confidence can be placed in its ability to predict future conditions. By adjusting input parameters and coefficients to conditions expected after implementation of the proposed environmental alteration, the behavior of the model provides a test of expected impacts that can be validated against mental predictions and data from other sources. In certain circumstances a simulation model may be the only way to predict impacts. In a large complex system it is often difficult even to hypothesize how an alteration to one component will affect the rest of the system. A simulation model may be the only way to assess the overall adjustments in the system.

MODEL CONCEPTUALIZATION

The process of objectifying mental images of a system into a diagrammatic or mathematical form is called model conceptualization. The structure and viewpoint engendered in the model should reflect the questions or specific objectives being addressed. The first step in this conceptualization process involves defining the scale or scope of the system to be modeled. Spatial and temporal boundaries must be drawn so as to define the system of study. We may, for instance, choose to define the boundaries of the Chesapeake Bay as its watershed up to the Pennsylvania border and down to the Bay Bridge-Tunnel. Or we may divide the Bay into subsectional regions (to be studied separately) according to long term mean salinity regimes, with a longitudinal slice between eastern and western shores. A general rule-of-thumb to follow in establishing boundaries is to select the largest scale of modeling at least one step broader than the level of the scientific or resource management question being addressed.

Once the extent of the system has been defined, we must define all the major inputs and outputs to and from that system across its boundaries. These trans-boundary flows should include exchanges of materials, energy, money and information. The system model is then further defined by identifying the important system components. These components should characterize the state of the system and involve some sort of storage of the things that flow through the system (material, energy, money or information). These components are, thus, often referred to as state variables.

A matrix of "connectedness" or "connectivity", with the model state variables as headings for both rows and columns, can next be developed to determine the degree to which the model components are explicitly interconnected. In general we find that, although we know that the system components are somehow interconnected, the degree of connectivity in this matrix is significantly less than 100% (Levins 1973). In fact, indirect connections are maintained between components by means of clustering into subsystems and hierarchies (Overton 1975). In modeling such complex systems this clustering characteristic can be implemented into model design so as to reduce the number of variables considered in the model. Thus, rather than modeling the behavior of all individual organisms in the Chesapeake Bay system, we can recognize some natural aggregations (i.e., populations, habitats, communities, etc.) and thus reduce our model's complexity. The exact level and angle of our conceptual slicing should be determined, again, according to the questions being addressed.

Hierarchies, with systems of subsystems of subsystems, etc., may exist in nature as mechanisms for decreasing complexity, and increasing control and stability (Weiss 1969; Simon 1973; Levins 1973). Similarly, we can design groups of models for these complex systems, each dealing with a different subsystem. This allows us tactically to view the vast complexity in an incremental fashion without losing the holistic perspective of the overall system. It must be remembered, however, that each time we aggregate a complex submodel into a single component of a given model of study, we lose resolution on the model's ability to mimic detailed dynamics of nature. As we aggregate mechanistic detail of interaction, model behavior becomes increasingly linear. System components should be aggregated together only if they have similar functional characteristics. Different biological populations, for instance, should be clustered together into a single model state variable only if they have similar characteristics of feeding, breeding and metabolic turnover-time.

Formal models are described with sets of mathematical equations. Statistical models can be used to represent empirically observed relationships between variables. However, statistical models can imply no cause-effect relationships between variables, whereas most scientific theory and certainly all resource management policies depend on causal inference. Sets of differential equations are used to indicate causal relationships and simulate behavior of natural systems. Hence, these are termed simulation models. Time-series phenomena can be addressed with these models, letting time be an independent variable in differential equations (that is, dynamics of system variables are described over time). Spatial patterns may also be modeled in detail using partial (or pseudo-partial) differential equations with each dimension in space also being an independent variable. Each equation describes the rate-of-change for a given variable over time (or space), and each term in the equation represents an interaction with another variable or with factors external to the model. Each term is a functional relationship between variables, the form of which is determined from results of scientific experiments.

SOME PRELIMINARY MODELS OF SUBMERGED AQUATIC VEGETATION IN THE CHESAPEAKE BAY

One hypothesis in this report is that the general decline in submerged aquatic vegetation of the Chesapeake Bay, is the result of some change (be it man-induced or natural) in the Bay's water quality. A secondary hypothesis might propose that certain human activities in the Bay's watershed had lead to these changes in water quality which were, in turn, responsible for the demise of SAV. If we were to suggest further that SAV communities provided important habitat, refuge and nutrition sources for important economic fisheries in the Bay, then we begin to see a clear conflict in natural resource use and a need for a rational management scheme. The conflict is, of course, between options of unmanaged use of the Bay for waste disposal versus use of the Bay as a source of productive fisheries. Couched within the framework of this management issue, we see a complex of interactions between human activities in the watershed region and the ultimate decline in economically important Bay populations.

To deal with this complex regional system in the detail needed for conflict resolution, we must develop our models of this system in a simplified, but realistic fashion. This is best done by building a hierarchical suite of models at several stages.

1) The largest scale model would include the entire watershed region consisting of natural and human systems, with both material and monetary exchanges between system components, focussing on the interactions between man and nature.

2) The next smaller scale might include the entire Chesapeake Bay as a large ecosystem with connections to the peripheral lands and between the various geographic sub-Bay systems (such as: brackish upper Bay; mid-salinity Bay; Chester and Choptank Rivers; Patuxent Rivers; Patuxent River; upper and lower Potomac River; Nanticoke River and Tangier Sound; upper and lower York and Rappahannock Rivers; mainstem saline Bay; and the upper and lower James River).

3) One of these geographic sub-Bay ecosystems could be considered as the next smaller scale--for example, the Patuxent River estuary. At this scale we would include in our model all the major ecosystem components aggregated by habitat and/or community type.

4) Further elaboration might include details on structure and function of one of these communities--for instance, to model the epifaunal or infaunal communities associated with SAV. Here we might consider predator-prey, competitive, commensal and symbiotic interactions between various animal populations in the community.

5) A fifth level of modeling detail might consider the population dynamics of one of the species potentially affected by SAV decline, such as blue crabs or striped bass. Such a model would include the important life stages or instars of the population (e.g., striped bass eggs, yolk sac larvae, finfold, post-finfold, young-of-year juveniles, large juveniles, mature adults) and such functional properties as consumption, respiration, production, mortality fecundity and recruitment.

6) Although there may be interest in many, successively smaller scales, the obvious basic ecological scale would involve models of whole-organism physiology. A pertinent model at this level would investigate the inhibitory characteristics of herbicides and silts on SAV, with plant organs or tissues being the basic components of this model.

A generalized diagrammatic model of the Chesapeake Bay watershed region is presented in Figure 46 .Here some of the main features of this region are depicted along with important relationships between human systems and Bay ecosystems. The regional economy is supported, in part, by local agriculture and fisheries, as well as manufacturing and selling commercial products in exchange for materials, services and fuels needed for economic operation. The natural processes of the Bay generate exploitable fisheries to help support the economy. In addition, the Bay provides other services to the economy such as transporting and assimilating wastes, providing recreational opportunities, and others--all of which contribute to the general image of the region associated with its ability to attract external economic investments. In this model we can see that any activities in the human systems which lead to runoff and discharge of wastes to the Bay can contribute (via food-chain and habitat modifications) to changes in the vitality of fisheries which help support that human economy. Implementation of various management policies could reduce any detrimental effects on one part of the economy (fisheries, as mediated through the Bay ecosystems) caused by activities at another part of the economy (farming, industries, etc.). Hence, to optimize the vitality of the regional economy we must understand the dynamic properties of the Chesapeake Bay ecosystems.

In a model of a sub-Bay ecosystem (such as the lower Patuxent River estuary) (Figure 47) we see that the ecosystem is split into two vertically differentiated halves (upper pelagic and lower benthic). This division is justified, in part, by vertical stratification which occurs during the summer. Riverflow and other runoff, as well as tidal currents, transports various materials and energies (including phosphate, nitrate, ammonia, dissolved and particulate organics, inorganic silt and salt, heat, herbicides, dissolved oxygen and carbon dioxide, phytoplankton and zooplankton) into and out from the system's water column. Those same water quality variables are stored temporarily in the ecosystems, during which time they interact with each other and with the rest of the system. Phytoplankton in the estuary consist of several important algal families and groups including diatoms, dinoflagellates, microflagellates, greens and bluegreens. They take up nutrients and absorb sunlight to generate photosynthate and oxygen. The kinetic and turbulent energies arising from river, wind and tidal forces produce vertical motions of advection and diffusion to transport water quality variables between upper and lower parts of the water column. The lower column water quality includes that which is contained in the sediment interstices.

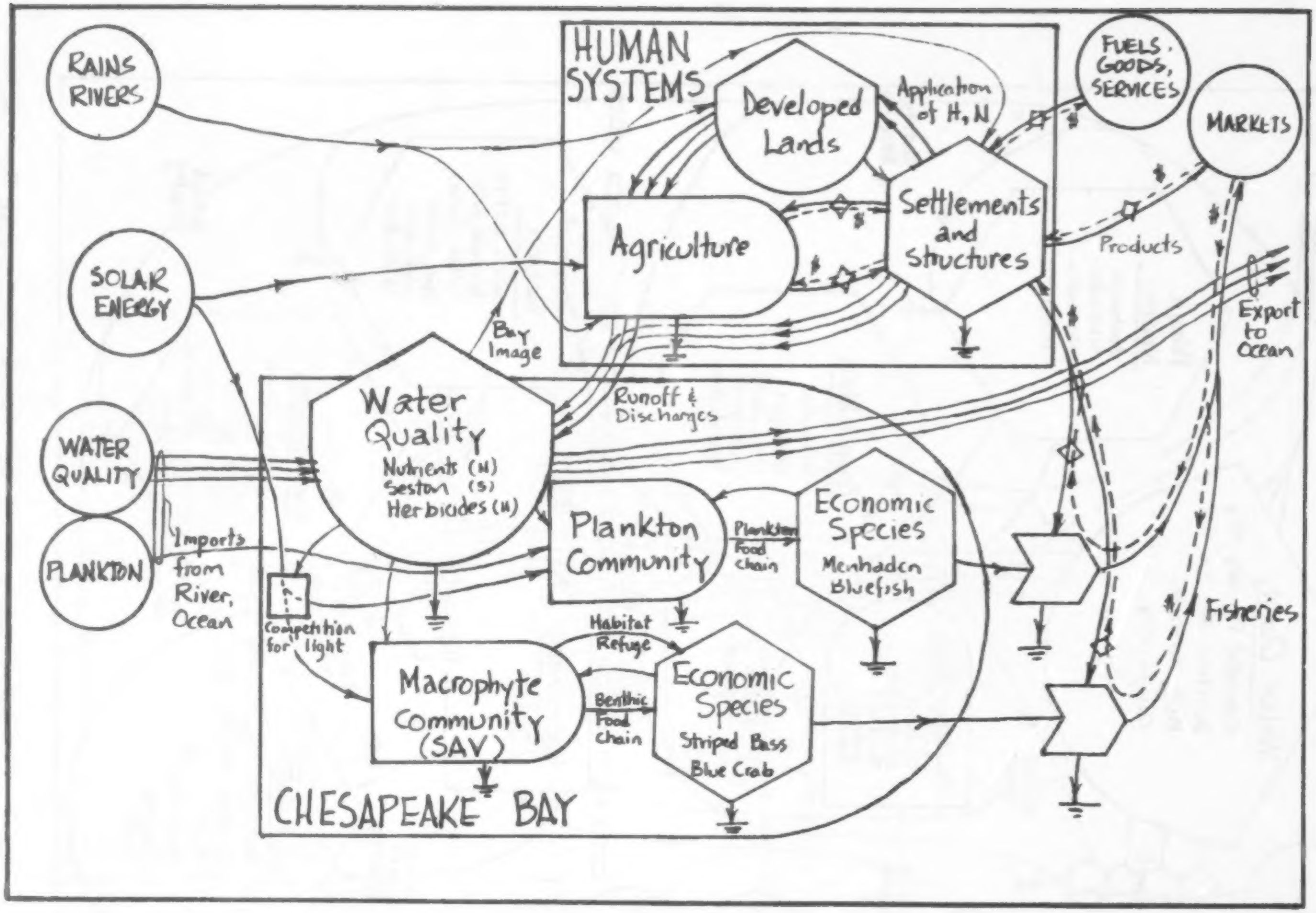
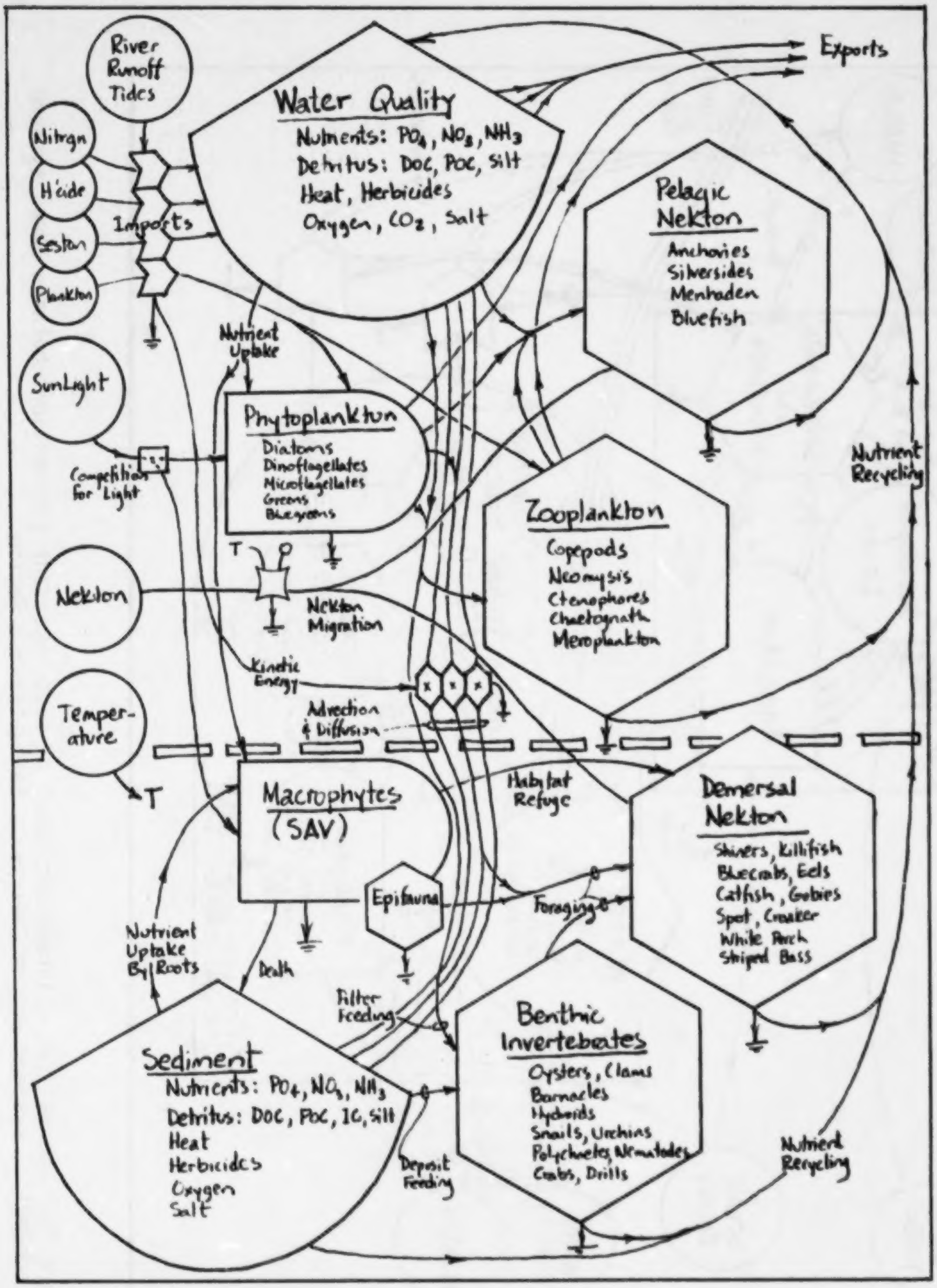

HUMAN SYSTEMS
RAINS RIVERS
SOLAR ENERGY
WATER QUALITY
PLANKTON
Imports from River, Ocean
Developed Lands
Application of H, N
FUELS, GOODS, SERVICES
MARKETS
Agriculture
Settlements and Structures
Products
$
Export to Ocean
Bay Image
Water Quality
Nutrients (N)
Seston (S)
Herbicides (H)
Runoff & Discharges
Plankton Community
Plankton Food chain
Economic Species
Menhaden
Bluefish
Fisheries
Competition for light
Habitat Refuge
Macrophyte Community (SAV)
Benthic Food Chain
Economic Species
Striped Bass
Blue Crab
CHESAPEAKE BAY

Figure 46. Generalized diagrammatic model of some important interactions between human systems and ecological systems of the Chesapeake Bay watershed region



River Runoff Tides
Exports
Water Quality
Nutrients: PO4, NO3, NH3
Detritus: DOC, POC, Silt
Heat, Herbicides
Oxygen, CO2, Salt
Nitrgn
H'cide
Seston
Plankton
Imports
Pelagic Nekton
Anchovies
Silversides
Menhaden
Bluefish
Nutrient Uptake
SunLight
Phytoplankton
Diatoms
Dinoflagellates
Microflagellates
Greens
Bluegreens
Competition for Light
Nutrient Recycling
Zooplankton
Copepods
Neomysis
Ctenophores
Chaetognath
Meroplankton
T
Nekton
Nekton Migration
Kinetic Energy
Advection & Diffusion
Temperature
T
Macrophytes (SAV)
Habitat Refuge
Demersal Nekton
Shiners, Killifish
Bluecrabs, Eels
Catfish, Gobies
Spot, Croaker
White Perch
Striped Bass
Epifauna
Foraging
Nutrient Uptake By Roots
Death
Filter Feeding
Benthic Invertebrates
Oysters, Clams
Barnacles
Hydroids
Snails, Urchins
Polychaetes, Nematodes
Crabs, Drills
Sediment
Nutrients: PO4, NO3, NH3
Detritus: DOC, POC, IC, Silt
Heat
Herbicides
Oxygen
Salt
Deposit Feeding
Nutrient Recycling

Figure 47. Model of a sub-estuary ecosystem of the Chesapeake Bay, major components and interactions

Benthic macrophytes (SAV) also take up nutrients and absorb sunlight for photosynthesis. However, in competition with phytoplankton SAV have the distinct disadvantage of growing from the bottom up (and thus having less access to light than phytoplankton which can be suspended near the water surface), but SAV have the potential advantage of having been able to take nutrients from the sediment waters, as well as from the water column. (McRoy and Barsdate 1970; McRoy et al. 1972; Best and Nantai 1977). Epifaunal animals living on the macrophyte structure, feed on epiphytes and detritus on the SAV leaves, keeping them clean and partly free of materials which block light from the leaves (Nagle 1968). Both SAV and phytoplankton death contribute to the organic deposits in the sediments. At high concentrations, various herbicides in the water column and sediments may inhibit photosynthesis of phytoplankton and SAV. However, phytoplankton have the potential advantage of being continuously seeded from external sources and having fast growth rates, so that if herbicides enter the system in pulses (Wu 1977) then phytoplankton can rejuvenate after a general mortality more quickly than SAV.

Zooplankton are similarly being continuously seeded from external sources. They filter phytoplankton and detritus from the water column and excrete nutrients which can then be recycled. Zooplankton consist of several different types distinguished by size, taxa, trophic habits or life cycle; and include both meroplankton and holoplankton, copepods and large neomysis, and carnivorous ctenophores and chaetognaths. Benthic invertegrates feed on detritus and phytoplankton filtered from the water column and on detritus deposited in the sediments. Their excretion and physical disruptive activities also keep nutrients cycling back to water column storages. These benthic infauna include filter-feeding types such as oysters, clams, barnacles, hydroids, and other polyps; deposit-feeders such as snails and worms; some herbivorous urchins and snails; and a few carnivorous crabs, drills and polychaetes. Nekton are grouped in this model according to main habitat type. The pelagic nekton include mostly planktivorous fish and their predators (bluefish). Demersal nekton include the forage fish which feed on benthic infauna and epifauna, as well as detrital matter, and their predators (striped bass). Both groups of fish migrate in and out of the ecosystem according to some seasonal cues (temperature, photoperiod, and etc.).

In Figure 48 the detailed mechanistic relationships between model components are explicitly shown. In this version of the sub-estuary model the level of aggregation is defined according to those variables considered most important to the overall ecosystem dynamics and to the basic issues addressed: 1) why have SAV declined, and 2) what is the effect of this decline on the dynamics of economically important populations? Some of the important features of this model not previously discussed in the more general version (Figure 47) are discussed below.

Temperature is an important interactive, external variable, involved in five metabolic functions in the system, and controlling five logic switches. Almost all of the heterotrophic components of the model are shown to have respiration and/or mortality as a function of density (i.e., state-variable interacting with itself in an outflow through a boxed-arrow). A large die-off of SAV through a switching function is designed to occur at the end of the growing season when temperatures go below a threshold. The deposited detritus is shown

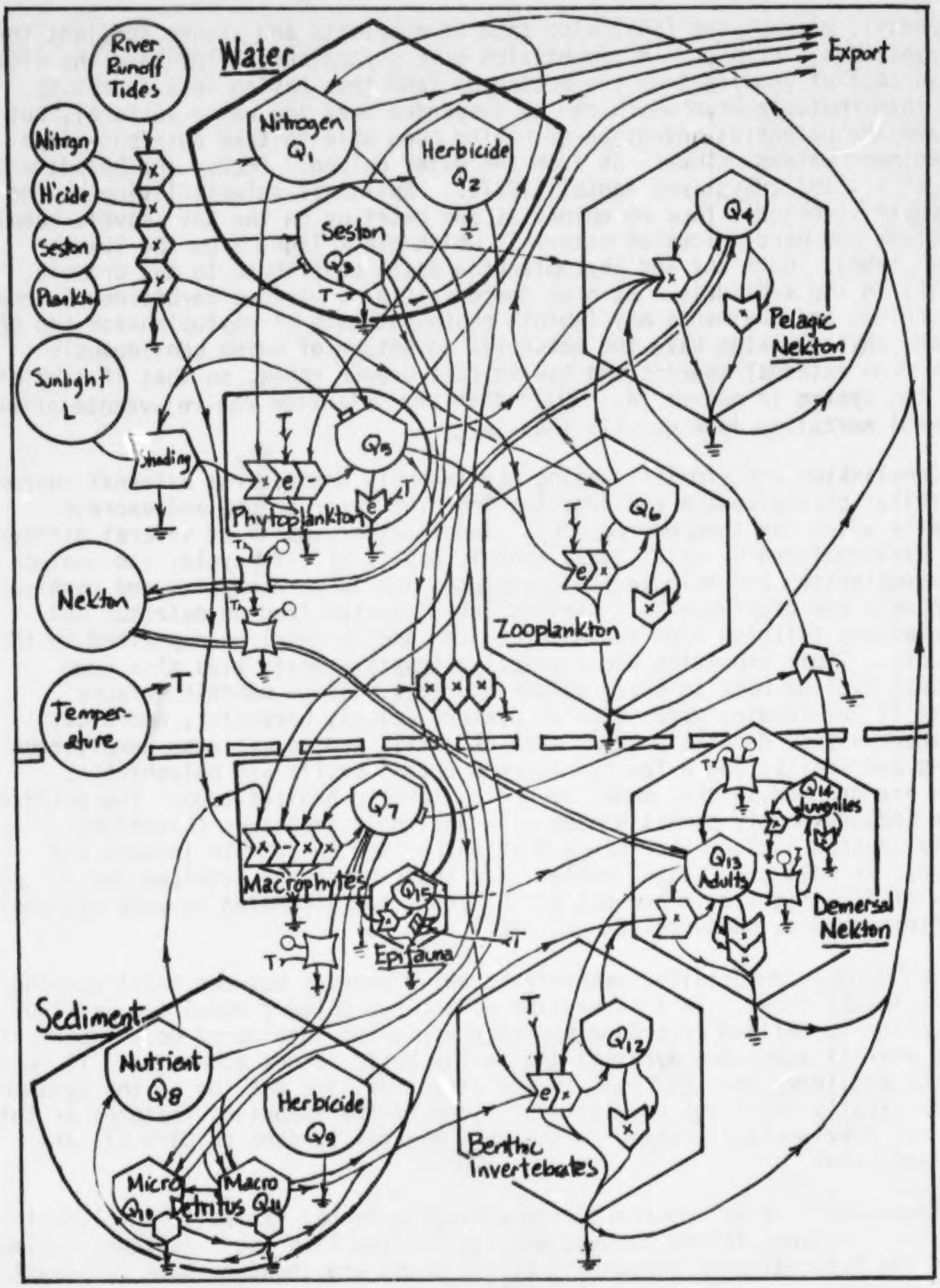

River Runoff Tides
Water
Export
Nitrogen Q_1
Herbicide Q_2
Nitrgn
H'cide
Seston
Plankton
Seston Q_3
Q_4
Pelagic Nekton
Sunlight L
Shading
Q_5
Phytoplankton
Q_6
Zooplankton
Nekton
Temperature
Q_7
Macrophytes
Q_{15}
Epifauna
Q_{14} Juveniles
Q_{13} Adults
Demersal Nekton
Sediment
Nutrient Q_8
Herbicide Q_9
Q_{12}
Benthic Invertebrates
Micro Q_{10}
Detritus
Macro Q_{11}

Figure 48. Model of a sub-estuary ecosystem of the Chesapeake Bay, mechanistic details appropriate for mathematical translation and eventual computer simulation

in two parts, separated by particle size, because of differences in time-constants and because the smaller pieces are known to be more nutritious (with more microbes per particle mass) (Odum 1970). A detail to the demersal nekton population structure is indicated with two separate life-stages considered. This was done because the juveniles of species such as white perch and striped bass are reported to depend on SAV for habitat (i.e. a refuge from predation) and for food, epifauna (e.g., Hollis 1952; Adams 1976a,b). SAV may thus affect recruitment success and overall population vitality.

A highly simplified model of a Chesapeake Bay sub-estuary is shown in Figure 49. This model was designed and simulated by W. Boynton and F. Lipschultz to investigate productivity and nutrient dynamics of the Bay, and we include it in this report merely as an example of some kinds of output that can be expected from ecosystem models such as that shown in Figure 47. The model was developed to examine the major influences on nutrient recycling in the upper Chesapeake bay. The model was found useful in providing insight into effects of summer stratification and fall turnover on ecosystem dynamics in the euphotoic zone. The effects of Susquehanna River flow on nutrient levels was also investigated with this model.

MATHEMATICAL EQUATIONS FOR MODEL SIMULATION

The time-varying behavior of an ecosystem model can be studied by transforming the initial conceptualization into mathematical terms. Each model state-variable has a differential equation to describe its dynamic character. Each input and output pathway to aid from the state-variable represents a single term in the equation for that variable, and each pathway represents some process which affects the variable. The terms in each equation may be linear (containing only one variable) or nonlinear (being a function of several variable and/or a power, exponential, logarithmic, etc. function of one variable). The differential equations are written with the change in quantity of each variable per unit change in time (or space) equal to the sum of all input and output terms. Since each term is a function of model variables themselves, which are changing with time, the entire model system exhibits a dynamic behavior.

Notice in Figure 48 that the model contains 15 state-variables, and thus needs 15 differential equations to be described. Similarly, in Figure 50 we see that there are five variables which need five equations for mathematical description. The equations used for these models are given in Tables 84, 85 and 86, respectively, with a general word description for each term. Terms in each equation which are controlled (on and off) by some external logic function (such as temperature and photoperiod thresholds to start fish migration) are indicated with a bracket above them. A dot above the symbol for a state-variable indicates the instantaneous-rate-of change for that variable, or in differential notation, dQ/dt. Pathway coefficients, indicated as $k_{i,j}$, represent the inherent characteristics in a pathway which are invariant within the time-and space-frame of the model. The "i" represents the donor or source of the flow (e.g. state-variable,(Q_i) and the "j" represents the recipient of the flow (Q_j). In general, subscripting provides a description of variables and pathways, while superscripting indicates power functions.

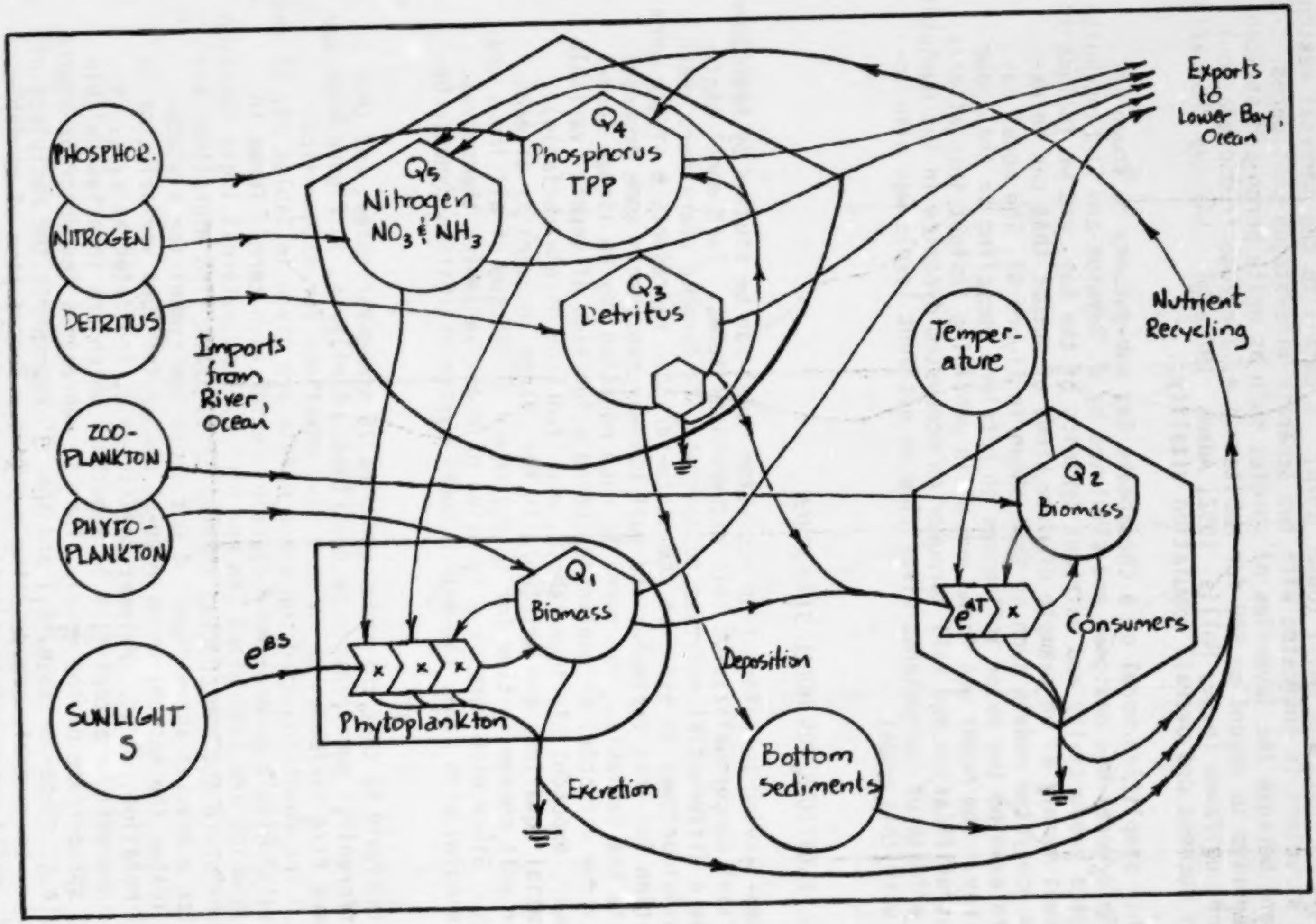

PHOSPHOR.
NITROGEN
DETRITUS
ZOO-PLANKTON
PHYTO-PLANKTON
SUNLIGHT S
Imports from River, Ocean
Q_5 Nitrogen NO_3 & NH_3
Q_4 Phosphorus TPP
Q_3 Detritus
Q_1 Biomass
e^{BS}
Phytoplankton
Excretion
Deposition
Bottom Sediments
Temper-ature
Q_2 Biomass
e^{aT}
Consumers
Nutrient Recycling
Exports to Lower Bay, Ocean

Figure 49. Simplified model of pelagic sub-estuary of Chesapeake Bay designed for preliminary studies of productivity and nutrient dynamics

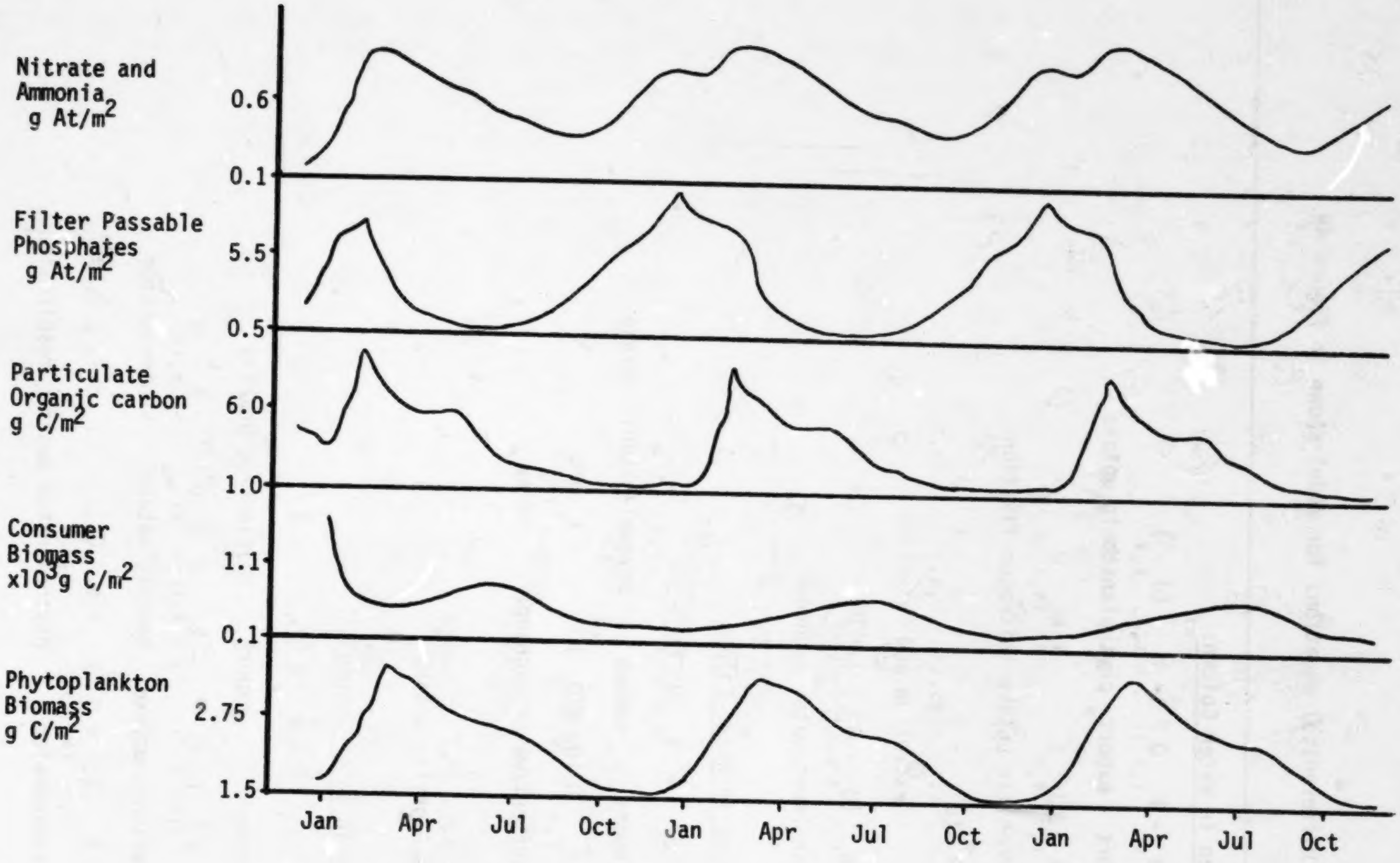


Figure 50. Sample output of simplified ecosystem model of the upper Chesapeake Bay

Table 84 . Differential equations for model shown in Figure 48.

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1) Nitrogen in Water Column:

$\mathring{Q} = J_1 - k_{1,0}Q_1F - k_{1,5}(J_{5,5})$

import export phytoplanktonic uptake

$- k_{1,7}(J_{7,7}) + k_{15,1}Q_{15}$

macrophytic uptake nitrogen fixation

$+ k_1 (J_{6,6} + J_{15,15} + J_{3,3} + J_{12,12})$

excretion and recycling

$+ k_{1,8}(Q_8 - Q_1) Q_{12} D$

sediment/water exchange

2) Herbicide in Water Column:

$\mathring{Q}_2 = J_2 - k_{2,0}Q_2 F - k_{2,5} (J_{5,5})$

import export phytoplanktonic uptake

$- k_{2,9} D (Q_9 - Q_2) - k_{2,2} Q_2$

sediment/water exchange decay

$- k_{2,7} (J_{7,7})$

macrophytic uptake

3) Seston in Water Column:

$\mathring{Q}_3 = J_3 - k_{3,0} Q_3 F - J_{3,6}$

import export zooplantkon grazing

$- k_{3,4} Q_3 Q_4 - k_{3,12}Q_3 Q_{12} e^{\alpha_{12}T} - k_{3,10} Q_3$

nektonic uptake benthic uptake sedimentation

$+ k_{10,3} (Q_{10} + Q_{11}) + k_{5,3} Q_5 - k_{3,3} Q_3 e^{\alpha_3 T}$

resuspension phytoplankton death respiration

4) Pelagic Nekton:

$$\overset{\circ}{Q}_4 = \underbrace{\left| J_{0,4} - J_{4,0} \right|}_{\text{seasonal migration}} + \underbrace{k_{3,4} Q_3 Q_4}_{\text{feeding on seston}}$$

$$+ \underbrace{k_{6,4} Q_6 Q_4}_{\text{feeding on zoopl.}} + \underbrace{k_{5,4} Q_5 Q_4}_{\text{feeding on nekton}}$$

$$- \underbrace{k_{4,4} (Q_4)^2}_{\text{respiration}} + \underbrace{J_{14,4}}_{\text{feeding on juveniles}}$$

5) Phytoplankton:

$$\overset{\circ}{Q}_5 = \underbrace{J_5}_{\text{import}} - \underbrace{k_{5,0} Q_5 F}_{\text{export}} - \underbrace{k_{5,3} Q_5}_{\text{death}}$$

$$- \underbrace{J_{5,6}}_{\text{zooplankton grazing}} + \underbrace{J_{5,5}}_{\text{primary productivity}}$$

$$- \underbrace{k_{5,16} Q_5 e^{\alpha_5 T}}_{\text{respiration}} - \underbrace{k_{5,4} Q_5 Q_4}_{\text{nektonic grazing}}$$

$$- \underbrace{k_{5,12} Q_5 Q_{12} e^{\alpha_{12} T}}_{\text{benthic grazing}}$$

6) Zooplankton:

$$\overset{\circ}{Q}_6 = \underbrace{J_6}_{\text{import}} - \underbrace{k_{6,0} Q_6 F}_{\text{export}} - \underbrace{k_{6,4} Q_4 Q_6}_{\text{nektonic predation}}$$

$$+ \underbrace{k_{6,6} (J_{3,6} + J_{5,6})}_{\text{grazing on phytopl. and seston}} - \underbrace{k_{6,6} (Q_6)^2}_{\text{respiration}}$$

7) Benthic Macrophytes (SAV):

$$\begin{aligned}\mathring{Q}_7 = {} & \underbrace{J_{7,7}}_{\text{primary productivity}} \quad \underbrace{- k_{7,16}\, Q_7\, e^{\alpha_7 T}}_{\text{respiration}} \\ & \underbrace{- \left\lceil k_{7,11} \quad Q_7 \right\rceil}_{\text{winter die off}} \quad \underbrace{- k_{7,15} Q_7 \quad Q_{15}}_{\text{herbivory}} \\ & \underbrace{- k_{17,11} \quad Q_7}_{\text{continual death}}\end{aligned}$$

8) Nitrogen in Sediments:

$$\begin{aligned}\mathring{Q}_8 = {} & \underbrace{k_{1,8}\,(Q_8 - Q_1)\; Q_{12}\; D}_{\text{sediment/water exchange}} \quad \underbrace{+ k_{10,8}(Q_{10})\, e^{\alpha_{10} T}}_{\text{micro-detri. respir.}} \\ & \underbrace{+ k_{11,8}(Q_{11})\, e^{\alpha_{10} T}}_{\text{macro-detritus respiration}} \quad \underbrace{- k_{8,7}\,(J_{7,7})}_{\text{macrophytic uptake}}\end{aligned}$$

9) Herbicide in Sediments:

$$\mathring{Q}_9 = \underbrace{k_{2,9}\; D\; (Q_9 - Q_2)}_{\text{sediment/water exchange}} \quad \underbrace{- k_{9,9}\; Q_9}_{\text{decay}}$$

10) Micro-Detritus in Sediments:

$$\begin{aligned}\mathring{Q}_{10} = {} & \underbrace{k_{12,10}\; (J_{11,12})}_{\text{break-down of macro-detritus by benthics}} \\ & \underbrace{- J_{10,12}}_{\text{benthic grazing}} \quad \underbrace{- k_{10,10}\; Q_{10}\; e^{\alpha_{10} T}}_{\text{respiration}} \\ & \underbrace{+ k_{13,10}\; (k_{11,13}\; Q_{11}\; Q_{13})}_{\text{break-down of macro-detritus by nekton}}\end{aligned}$$

11) **Macro-Detritus in Sediments:**

$\mathring{Q}_{11} = \overline{|k_{7,11} \quad Q_7|}$

winter macrophytic die-off

$+ k_{17,11} \quad Q_7$

continual macrophytic death

$- k_{11,11} \quad Q_{11} \quad e^{\alpha_{11} T}$

respiration

$- J_{11,12}$

benthic grazing

$- k_{11,13} \quad Q_{11} \quad Q_{13}$

nekton grazing

12) **Benthic invertebrates:**

$\mathring{Q}_{12} = (J_{3,12} + J_{5,12})$

suspension-feeding

$- k_{12,13} \quad Q_{12} \quad Q_{13}$

nektonic predation

$+ (J_{10,12} + J_{11,12})$

deposit - feeding

$- k_{12,12} \quad (Q_{12})^2$

respiration

13) **Demersal Nekton, Adults:**

$\mathring{Q}_{13} = \overline{|J_{0,3} - J_{3,0}|}$

seasonal migration

$+ k_{11,13} \quad Q_{11} \quad Q_{13}$

feeding on detritus

$- \overline{|k_{13,14} \quad Q_{13}|}$

spawning

$+ k_{12,13} \quad Q_{12} \quad Q_{13}$

feeding on benthics

$+ \overline{|k_{14,13} \quad Q_{14}|}$

recruitment

$- k_{13,13} \quad (Q_{13})^3$

respiration

14) **Demersal Nekton, Juveniles:**

$\mathring{Q}_{14} = k_{15,14} \quad Q_{14} \quad Q_{15}$

feeding on epifauna

$- k_{14,14} \quad (Q_{14})^2$

respiration

$- J_{14,4}$

predation by nekton

$+ \overline{|k_{13,14} \quad Q_{13}|}$

spawning

$- \overline{|k_{14,13} \quad Q_{14}|}$

recruitment

15) Epifaunal Community:

$$\overset{\circ}{Q}_{15} = k_{7,15} \; Q_7 \; Q_{15} \qquad - k_{15,14} \, Q_{14} \, Q_{15}$$

grazing epiphytes — predation by nekton

$$- k_{15,5} \; (Q_{15})^2 \; e^{\alpha_{15} T}$$

Table 85. Special functions used in equations given in Table 84.

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1) Primary Productivity, Phytoplankton:

$$J_{5,5} = k_{5,5} \underbrace{(e^{\alpha_{5,5} T})}_{\text{temperature}} \underbrace{\left(\frac{Q_1}{1 + a_5 Q_1}\right)}_{\text{nitrogen}} \underbrace{\left(\frac{L}{1 + b_5 L}\right)}_{\text{sunlight}} \underbrace{(1 - C_5 e^{\beta_5 Q_{12}})}_{\text{herbicide inhibition}}$$

2) Primary Productivity, Macrophytes:

$$J_{7,7} = k_{7,7} \underbrace{(e^{\alpha_7 T})}_{\text{temperature}} \underbrace{\left(\frac{(Q_1 + Q_8)}{1 + a_7 (Q_1 + Q_8)}\right)}_{\text{nitrogen}} \underbrace{\left(\frac{L}{1 + b_7 L}\right)}_{\text{sunlight}} Q_7 \underbrace{(1 + d_7 Q_{15})}_{\text{epifaunal cleaning}}$$

$$\underbrace{(1 - C_7 e^{\beta_7 Q_2})}_{\text{herbicide inhibition}}$$

3) Shading of Sunlight:

$$L = L_0 (K - e^{\alpha_0 (Q_3 + Q_5)})$$

4) Zooplankton Grazing on Seston:

$$J_{3,6} = k_{3,6}(e^{\alpha_3 T}) Q_6 \left(\frac{Q_3}{1 + k_3 Q_3}\right)$$

5) Zooplankton Grazing on Phytoplankton:

$$J_{5,6} = k_{5,6}(e^{\alpha_3 T}) Q_6 Q_5$$

6) Benthic Invertebrates Feeding, on Phytoplankton:

$$J_{5,12} = k_{5,12} Q_5 e^{\alpha_{12} T} (Q_{12} + \underbrace{d_{12} Q_{13}}_{\text{population management}})$$

7) Benthic Invertebrates Feeding, on others:

$$J_{i,12} = k_{i,12} Q_i e^{\alpha_{12} T} (Q_{12} + d_{12} Q_{13})$$

where i is food item:
Macro-Detritus, i = 11
Micro-Detritus, i = 10
Seston, i = 3

Table 86. Differential equations for model shown in Figure 49.

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1) Phytoplankton:

$$\overset{\circ}{Q}_1 = \underbrace{k_{1,1}\, Q_1\, Q_4\, Q_5\, L}_{\text{producitivity}} - \underbrace{0.125\, Q_1}_{\text{death}} - \underbrace{k_{1,2}\, Q_1\, Q_2\, e^{.75T}}_{\text{predation}}$$

$$- \underbrace{k_{1,6}\, Q_1}_{\text{export}} - \underbrace{0.24\, Q_1}_{\text{respiration}} + \underbrace{0.0006}_{\text{import}}$$

2) Consumers:

$$\overset{\circ}{Q}_2 = \underbrace{k_{2,1}\, Q_1\, Q_2\, e^{.75T}}_{\text{feeding on phytopl.}} + \underbrace{k_{2,3} Q_3 Q_2 e^{.75T}}_{\text{feeding on detritus}}$$

$$- \underbrace{0.25\, Q_2}_{\text{export and death}} + \underbrace{0.0009}_{\text{import}}$$

3) Detritus:

$$\overset{\circ}{Q}_3 = \underbrace{J_{0,3}}_{\text{river input}} + \underbrace{0.125\, Q_1}_{\text{phytopl. death}} + \underbrace{k_3 (k_{1,2} Q_1 + K_{3,2}\, Q_3)\, Q_2\, e^{.75T}}_{\text{consumer excretion}}$$

$$- \underbrace{k_{3,7}\, Q_3}_{\text{sedimentation}} - \underbrace{k_{3,6}\, Q_3}_{\text{export}} - \underbrace{k_{3,2}\, Q_2\, Q_3\, e^{.75T}}_{\text{consumer grazing}}$$

4) Phosphate:

$$\overset{\circ}{Q}_{14} = \underbrace{J_{0,4}}_{\text{river input}} + \underbrace{J_{8,4}}_{\text{sediment regeneration}} + \underbrace{k_{1,4}\, (.125) Q_1}_{\text{phytopl. excretion}}$$

$$+ \underbrace{k_{3,4} Q_3}_{\text{detritus regeneration}} - \underbrace{k_{4,1} Q_1 Q_4 Q_5 L}_{\text{phytoplankton uptake}} - \underbrace{k_{4,6}\, Q_4}_{\text{export}}$$

5) Nitrogen:

$$\dot{Q}_5 = \underset{\text{river input}}{J_{0,5}} \quad + \underset{\text{detritus regeneration}}{k_{3,5}\,Q_3} \quad + \underset{\text{phytopl. excretion}}{k_{1,5}(.125Q_1)}$$

$$\underset{\text{consumer excretion of ammonia}}{+ k_{2,5}(k_{2,1}\,Q_1\,Q_2\,e^{.75T} \quad + k_{2,3}\,Q_2\,Q_3\,e^{.75T}}$$

$$\underset{\text{sediment regeneration}}{+ J_{8,5}} \quad \underset{\text{export}}{- k_{5,6}\,Q_5} \quad \underset{\text{phytopl. uptake}}{- k_{5,1}\,Q_1\,Q_4\,Q_5\,L}$$

CALIBRATION OF MODEL

To make the model operational, pathway coefficients involved in each term of each equation must be numerically evaluated or calibrated. Each equation term represents a flow of material, energy, or information which is stated as a function of one or more model state-variables and one (occasionally more) coefficient that characterizes inherent properties of the pathway. The model is calibrated by estimating the values of these coefficients. If data are available for a small time segment to establish the actual flow, as indicated by a given equation term, and the mean levels of each state-variable involved in that term, then the coefficient for the term can be algebraically calculated. Numerical values must also be assigned for the model's external forcing functions. Here, such factors as sunlight and temperature, which are external to the boundaries defined for the model in Figure 49 are described either by discontinuous time-series of input data or by some continuous function fit to that data. With model coefficients and forcing functions numerically defined, and with initial values (starting points) assigned to each state-variable, the model is ready to be programmed and simulated on a computer.

If we were to try to calibrate and simulate the model given in Figure 48 and Tables 84 and 85 for a particular sub-basin of the Bay, we would first seek all available information describing the structural and functional parts of that sub-Bay contained in the model. For the Patuxent estuarine ecosystem considerable amount of such data is available, and is summarized in Table 87. Comparing the variables listed in this table against the variables given in the model equations cf Table 84 (where each Q is a state-variable and each equation term is a flow variable), we find that the available data fills about 15 percent of the data needed to calibrate this model. This disparity between needs and availability helps to demonstrate major lacunae in the general state of our knowledge about this system. In Table 88 several additional pieces of data pertinent to the model are provided mostly from other similar ecosystems. The number of data values needed to completely calibrate this model can be reduced by the number of equations (15 in this case) by carefully selecting the time for calibration from the time-series of data, and by making a simple assumption of steady-state. If we calibrate the model from mid-winter data, for instance, we can assume that most ecological state-variables are at or near a minimum value. Thus, their rate-of-change is close to zero. By assuming that these rates-of-change are equal to zero, we can estimate one term in the equation if we know all the others. An example will illustrate the process of calculating coefficients. In equation 6 of Table 84 we find that zooplankton respiration is represented by $k_{6,6}(Q_6)^2$. In August zooplankton biomass was $.025 gC/m^2$ and respiration was $.005\ gC/m^2d$, and we thus calculate the respiration coefficient $k_{6,6} = (.005)/(.025)^2 = 8.0$. (Table 79).

SIMULATION OF MODEL

Once the model is fully calibrated it can be numerically simulated on an analog or digital computer. Initial simulation runs involve fine-tuning adjustments of coefficients to match model behavior with field data. In some cases, where no amount of coefficient adjustment can improve the fit of model to data,

Table 87. Some data from related ecosystems useful in calibrating a model of Patuxent estuarine ecosystem

Variable	Mean Value	Ecosystem	Reference
Rate of Sediment Deposition	14 mm/yr	Coastal Eelgrass	Thayer et al. (1975)
Carbon Content of Sediment	1.8%	Coastal Eelgrass	Thayer et al. (1975)
Carbon/Nitrogen of Sediment	14	Patuxent Estuary	Flemer et al. (1970)
Epifaunal Biomass	.47gC/m^2	Coastal Eelgrass	Thayer et al. (1975)
Epifaunal Assimilation	.018gC/m^2/d	Coastal Eelgrass	Thayer et al. (1975)
Epifaunal Respiration	.014gC/m^2/d	Coastal Eelgrass	Thayer et al. (1975)
Benthic Invertebrate Biomass	3.31gC/m^2	Coastal Eelgrass	Thayer et al. (1975)
Benthic Assimilation	.06gC/m^2/d	Coastal Eelgrass	Thayer et al. (1975)
Benthic Respiration	.05gC/m^2/d	Coastal Eelgrass	Thayer et al. (1975)
Sediment Detrital Respir.	.62gC/m^2/d	Georgia Estuary	Smith (1971)
Demersal Fish Biomass	.78gC/m^2	Coastal Eelgrass	Adams (1976a,b)
Demersal Fish Respiration	.02gC/m^2/d	Coastal Eelgrass	Adams (1976a,b)
Demersal Fish Consumption	.03gC/m^2/d	Coastal Eelgrass	Adams (1976a,b)
Macrophyte Biomass	230gC/m^2	Eelgrass	McRoy (1970a)
	53gC/m^2	Eelgrass	Penhale (1977)

Table 88. Summary of monthly data available for calibrating a model of Patuxent estuarine ecosystem as a subestuary of the Chesapeake Bay

Variable	Unit	Month of year											
		J	F	M	A	M	J	J	A	S	O	N	D
Temperature[a]	^oC	3.5	2.0	3.5	15.0	23.5	26	28	29	27	19	8.5	4.0
Extinction Coefficient[a]	-	2.0	2.9	3.5	3.0	3.0	3.0	4.5	6.5	3.5	2.5	4.5	2.0
Community Gross Prod.[b]	$gC/m^2/d$	-	1.0	0.8	2.5	3.0	5.3	4.8	7.5	4.0	1.5	0.3	-
Community Respiration[b]	$gC/m^2/d$	-	1.5	0.8	6.5	7.5	5.3	6.0	7.0	12.0	3.0	2.0	-
Phytopl. Net Prod.[c]	$gC/m^2/d$	-	-	-	-	0.3	0.5	0.8	1.1	.95	0.9	0.8	-
Phytoplankton Biomass[c]	gC/m^2	-	-	-	-	2.0	1.8	1.7	1.2	1.3	1.2	1.0	-
Zooplankton Respir.[c]	$gC/m^2/d$	-	-	-	-	.005	.009	.006	.005	.002	.0005	.0005	-
Zooplankton Biomass[c]	gC/m^2	-	-	-	-	.05	.06	.03	.025	.02	.015	.01	-
Zooplankton Production[c]	$gC/m^2/d$	-	-	-	-	.03	.05	.025	.02	.015	.01	.002	-
Inorganic Nitrogen[a]	gN/m^2	.90	.72	.78	.50	.08	.12	.08	.40	.16	.34	.54	.52
Seston[a]	g/m^2	64	90	140	70	110	90	110	120	140	100	80	70

[a] Data for 1968-1969 (Flemer, Hamilton, Keefe and Mihursky 1970).
[b] Data for 1969 (Cory 1974).
[c] Data for 1970 (Heinle 1974; Flemer 1970).

the equations themselves must be modified according to alternative mechanisms of interaction which correspond to documented facts or reasonable hypotheses. After the model is adjusted to match the data from which it was calibrated, its behavior should be "varified" or validated" by comparison with an independent data set. Any modifications needed to meet criteria of "reasonable validation" must involve changes in model equations rather than coefficients, so that model characteristics can be made to match both data sets. During the process of model verification, confidence is built in the general ability of the model to have realistically. Eventually, then, the model would be used to investigate system responses to modified external forcing functions. Thus, we might want to forecast the response of our Chesapeake Bay ecosystem model to increased turbidity, nutrients, and/or herbicides. We might remove SAV from the model and observe how the ecosystem (particularly fisheries) reacts. Predictive simulations allow the modeler to investigate ecosystem responses to a spectrum of different conditions, and compare these responses to field observations to test causative factors for ecosystem modifications.

The simplified ecosystem model of the upper Chesapeake Bay described in Figure 49 and Table 86 was simulated in some preliminary computer runs. Data for phytoplankton biomass, detritus, light and temperature were taken (Table 88) from Flemer (1970), while data for the consumer state-variable (comprised primarily of zooplankton) was taken from Heinle (1974) and nutrient concentrations and dynamics were based on the work of Whaley et al (1966). In Figure 50 we see that levels of nitrogen and phosphorus increase during the late fall and winter as a result of microbial mineralizariom and lack of autotrophic uptake. Decomposition of detritus which is carried by the river, further increased nitrogen and phosphorus concentrations. These increases, along with higher levels of insolation, permitted phytoplankton biomass to reach a peak in April and May. Consumers show less seasonality probably due to their ability to switch their feeding between detritus and phytoplankton according to which is more abundant. Consumer standing stocks reach an annual maximum in July, lagging about three months behind peak phytoplankton and zooplankton during winter as a result of decreased sunlight and temperature, respectively.

This model is presented here merely to demonstrate some methodologies involved in the simulation process. More detailed models (such as shown in Figure 48 at several levels of focus are needed for addressing questions of SAV interactions in the Chesapeake Bay.

CHAPTER 6

POSSIBLE MANAGEMENT OPTIONS

Until further experimental and field research is completed, it is difficult to determine what factor or combination of factors are actually responsible for the submerged aquatic declines in the Chesapeake Bay. Therefore, any suggestions of best management practices must be considered tentative at this time.

If the point source discharges from sewage and water treatment plants are determined to constitute a major problem, tertiary treatment of wastewater may help to some extent in relieving the high nutrient burden on SAV communities. However, chlorination problems would still be present. Chlorination alternatives that could be implemented immediately before discharging wastewater into the estuary should be investigated.

One possibility is to pass the effluent from existing sewage treatment plants under ultra-violet radiation sources to kill remaining pathogens before it is discharged. This might greatly reduce the need for chlorination. Another possibility is to use land treatment as a final biological filter to eliminate pathogens from the water before it reaches the Chesapeake Bay. Studies now under way at St. Charles City, Maryland show that forested land can act as an effective buffer in regard to bacteria. More extensive studies pioneered at Pennsylvania State University have found similar results.

Such land disposal methods are limited primarily to low density areas where land is both available and not prohibitive in cost. Around major cities such as Washington, D. C. and Baltimore it may not be possible to have both submerged aquatic vegetation and high chlorination and nutrient levels due to the necessity of large sewage and water treatment plants.

Further management options deal with high turbidities due to sedimentation. There is evidence that turbidity levels are lower in the 1970s (see Chapter 4). This may be due to ever increased effectiveness at controlling soil erosion around the perimeters of the Chesapeake Bay estuary. The use of no-till agriculture may be a prime reason for the decreased turbidity that seems to be apparent in many of the Bay's subestuaries during the 1970s. More work should be done to evaluate the effects of true no-till agriculture in regard to erosion control. By contrast to erosion from runoff, wind erosion in the drier years of 1976 and 1977 seems to be more prevalent. Encouraging farmers to plant wind breaks along fields would help alleviate this situation.

Another problem which many areas of the Bay are experiencing is shoreline erosion. The costs of arresting this natural process are so staggering that such measures may be prohibitive on a large scale. Also since the Chesapeake Bay has been experiencing shoreline erosion and subsequent sedimentation since its formation in Pleistocene, these processes cannot alone account for SAV losses.

More important than natural erosion processes are large construction projects and stream channelization efforts which are often responsible for dumping tremendous loads of sediments into the Bay estuary. A Bay area example of such a project is the "stream improvement" dredging being conducted by the U.S. Department of Agriculture Soil Conservation Service in the Delaware portion of the Choptank River. While appearing to be cost effective in Delaware because of an increase in usable agricultural acreages, the downstream effects may prove to be devastating not only to SAV but also to important commercial faunal species.

The possibility exists that significant leakage of new agro-chemicals into streams and tributaries may be the cause of Baywide declines in SAV. If particular herbicides utilized in the Bay area prove to be a major problem, it may be possible to substitute other available chemical compounds which are less harmful to SAV. Before any substitutions are made, careful screening of the existing compounds needs to be implemented at several levels. Submerged aquatics (including their symbionts) should be bioassayed and Bay ecosystem responses need to be determined before specific suggestions can be made.

Another management option might be to encourage area farmers to construct ditches that lead to small ponds where complete biodegradation of chemicals would occur before draining into the estuary. These small ponds would also act as sediment traps. No changes in "minimum or no till" agriculture are suggested until a thorough understanding of their benefits and environmental costs are tabulated, using economic and ecological modeling approaches.

Another possibility is to reestablish SAV populations using planting techniques. These techniques are currently being tested by the Maryland Department of Natural Resources (Stotts, pers. com.). Thus far there seems to be considerable difficulty in establishing large beds. This may help support one of the key conclusions in this study: that the decline in submerged aquatics is related to some overall change in water quality of the Bay, rather than to some intrinsic cyclic population phenomenon of the Bay ecosystem. Until we understand the dynamics of the SAV decline and its implications on the Chesapeake Ecosystem, it may not pay to attempt large scale replanting.

A more reasonable approach might be to incorporate existing productive beds of submerged aquatics into an estuarine sanctuary. These could then be protected from clam dredges and small boat propellers which have been suggested as negatively impacting the submerged aquatic populations. This would also provide areas where researchers could set up long term experiments which need to be done to answer our management questions.

One final management option which should be considered is the control of grazing waterfowl in selected sanctuary areas. This might be especially helpful in the case of mute swans which have the potential of consuming large amounts of submerged macrophytes. There is little evidence that the mute swans have been important in the past SAV decline. However, if the population continues to increase at its present rate of growth it could devastate remaining healthy populations. Thus, the Maryland Department of Natural Resources ought to consider opening a mute swan hunting season in the future in areas where SAV beds are being overgrazed. Since this species does not appear to be a problem in Virginia, no action needs to be considered there at this time.

In conclusion, however, because of the complexity of issues involved in the submerged aquatic decline in the Bay, implementation of any of the above management options is premature. Before any decisions can be made, regional analysis and modelling techniques (described in Chapter 5) need to be applied to the problem. Using these tools it should be possible to determine which management alternatives provide optimal utlization of natural resources in the Bay as well as insure the high productivity of the agricultural and industrial activities in the surrounding Chesapeake watershed.

CHAPTER 7

ANNOTATED BIBLIOGRAPHY OF PUBLISHED MATERIAL RELATED TO SUBMERGED AQUATIC VEGETATION IN THE CHESAPEAKE BAY

Anderson, R.R. 1966. Plant ecology of the upper Patuxent River estuary with special reference to the effects of thermal pollution on macrophytes. Ph.D. thesis, Univ. Maryland. 99pp.

The effects of temperature changes due to effluent from the Potomac Electric Power Company's Chalk Point generating plant were studied from June 1963 to June 1966. *Ruppia maritima* was found to have declined around the effluent canal and *Potamogeton perfoliatus* increased in coverage in the same area.

Anderson, R.R. 1969. Temperature and rooted aquatic plants. Chesapeake Sci. 10(3 and 4):157-164

The effects of temperature on respiration and photosynthesis of aquatic plants were studied using a Gilson differential respirometer. An historical review of research and present research by the author is presented.

Anderson, R.R. 1970. The submerged vegetation of Chincoteague Bay, pp. 136-155. *In* Assateague ecological studies. Univ. Maryland CBL Ref. No. 446.

Hydrophytes are important to the marine environment because of their soil binding roots, foliage which provides food and shelter for marine fauna, etc. In the Chincoteague Bay area, *Zostera marina* (eelgrass) and *Ruppia maritima* (widgeongrass) are the two dominant submerged aquatic species. A two-year study was conducted with emphasis on (1) type and distribution and (2) evaluation of primary production of species. Recommendations are presented for future dredging operations.

Anderson, R.R., R.G. Brown and R.D. Rappleye. 1965. Mineral composition of Eurasion water milfoil, *Myriophyllum spicatum* L. Chesapeake Sci. 6(1):68-72.

Material was collected from June 1962 to January 1963 to determine feasibility of milfoil use as a commercial fertilizer. Specimens collected at water temperatures from 0.2 to 30.0 C, pH values from 5.8 to 9.5 and salinities from 1.07 to 16.4ppt. Results indicate low N-P-K values that would not be economically feasible for commercial fertilizers.

Anderson, R.R., R.G. Brown and R.D. Rappleye. 1968. Water quality and plant distribution along the Upper Patuxent River, Maryland. Chesapeake Sci. 9(3):145-156.

From June 1963 to June 1966, a study was made of the Upper Patuxent River, Maryland, to determine the distribution of submerged and emergent aquatic vegetation. The boundary between fresh and saline water was found to be 0.3 ppt. Fluctuations over a 24-hour period reflected biological activity and tidal changes.

Anderson, R.R. 1972. Submerged vascular plants of the Chesapeake Bay and tributaries. Chesapeake Sci. 13(suppl):S87-S89.

A generalized summation of the present knowledge of submerged aquatic plants of the Chesapeake Bay and its tributaries including taxonomy, distribution and abundance, biology, ecosystems and pollution.

Anonymous. 1959. Chesapeake, stronghould of blue crab fishery. Nat. Fisherman. 40:13,30-31.

Maryland supplies 66% of the entire U.S. blue crab harvest. The crab industry of Maryland including the types, methods of catching and the crab life cycle are briefly discussed.

Bayley, S., H. Rabin, and C.H. Southwick. 1968. Recent decline in the distribution and abundance of Eurasian milfoil in Chesapeake Bay. Chesapeake Sci. 9(3):173-181.

Eurasian water milfoil declined in 1965 to 1967 in Chesapeake Bay. Detailed mapping of milfoil occurrence was done in Middle, Back and Rhode Rivers in July and September of 1966 and 1967. The decline exceeded 95%, and was associated with Northeast disease and Lake Venice disease.

Bayley, S., V.D. Stotts, P. Springer and J. Steenis. In press. Changes in submerged aquatic macrophyte populations at the head of the Chesapeake Bay, 1958-1975.

Changes due to large inputs of nutrients and silts, man's influence, epidemics and waterfowl population fluctuations are discussed in relation to submerged aquatic vegetation at the head of the Chesapeake Bay. The biological wave phenomena is also considered.

Bean, G.A., M. Fusco, W.L. Klarman. 1973. Studies on the "Lake Venice Disease" of Eurasian milfoil in the Chesapeake Bay. Chesapeake Sci. 14:279-280.

Samples of healthy and diseased *Myriophyllum spicatum* were taken from the Rhode River in 1972 to determine the way in which the "Lake Venice Disease" was transmitted and the causal agent. It was shown that the disease was only transmitted under stress from low light intensity and direct inoculation.

Beaven, G.F. 1960. Temperature and salinity of surface water at Solomons, Maryland. Chesapeake Sci. 1(1):2-11.

Daily temperature and salinity values averaged for a 20-year period, 1938-1957, and compared with other parts of the Chesapeake Bay. Water temperatures varied from 31 to 0.8 C. Seasonal means are: winter 4.3 , spring 11.9 C , summer 25.6 C and fall 18.2 C. Extreme salinity values are 20.4 and 5.4 ppt. Seasonal means are: winter 14.8, spring 11.4, summer 12.3 and fall 15.7 ppt.

Beaven, G.F., C.K. Rawls and G.E. Beckett. 1962. Field observations upon estuarine animals exposed to 2,4-D. Proc. Northeast Weed Control Conf. 16:449-458.

A study to determine if 2,4-D used in the control of Eurasian water milfoil is lethal to oysters, crabs, clams and fish. From the data gathered, 2,4-D in attaclay pellets was found to be non-toxic to crabs, toxic in varying degrees to fish and a serious threat to oysters, clams and other bottom organisms.

Bellrose, F.C. 1976. Ducks, geese and swans of North America. Stackpole Books, Harrisburg, Pa. 543pp.

Water species identification, population status, distribution, breeding and food habits in North America are defined along with species habitat and food resources of the Chesapeake Bay.

Brady, D.K. 1976. Are the Chesapeake Bay waters warming up? Chesapeake Sci. 17(3):225-227.

Water temperature records from various stations on the Chesapeake were correlated to determine the possibility of a warming trend. Because of insufficient data no definite trend was indicated. The periodic fluctuations seemed to be random.

Chamberlain, E.B., Jr. 1948. Ecological factors influencing the growth and management of certain waterfowl food plants on Back Bay National Wildlife Refuge.Thirteenth North American Wildlife Conf. pp. 347-356.

Since the early 20th century, waterfowl food plants on Back Bay National Wildlife Refuge have been studied. Factors influencing food plant production are salinity, hydrogen ion concentration, dissolved oxygen, water temperature and turbidity.

Clark, L.J., D.K. Donnelly and O. Villa, Jr. 1973. Nutrient enrichment and control requirements in the upper Chesapeake Bay. Summary and conclusions from forthcoming Tech. Rept. 56. EPA-903/9-73-002-a. 92pp.

Series of conclusions and graphically displayed supportive data relevant to the current eutrophication problem in the upper Chesapeake Bay. Phosphorus and nitrogen from the Susquehanna River Basin and the Baltimore metro area are determined as to maximum allowable loadings.

Clark, L.J., V. Guide and T.H. Pheiffer. 1974. Nutrient transport and accountability in the lower Susquehanna River basin. Tech. Tept. 60. EPA-903/9-74-014. 91pp.

One-year conprehensive nutrient study in the lower Susquehanna River basin to determine: (1) average and seasonal variations in nitrogen and phosphorus loadings; (2) delineation of point and non-point source nutrient contributions; (3) fate of nutrients in impounded areas and (4) seasonal mass balance of nutrient loadings.

Davis, J. ed. 1974. The effects of tropical storm Agnes on the Chesapeake Bay estuarine system. Rept. U.S. Army Corps of Engineers, Baltimore. DACW 31-73-C-0189.

Analysis of the effects of tropical storm Agnes on the hydrology, geology, water quality, biology and economics of the Chesapeake Bay.

Haven, D.S. 1963. Mass treatments with 2,4-D of milfoil in tidal creeks in Virginia. Proc. Southern Weed Control Conf. 16:345-350.

Field tests were performed in 1962 using 2,4-D on *Myriophyllum spicatum* in the Lower Machodoc Creek, Virginia. Pelleted formulations of 2,4-D were found to effectively control milfoil in tidal creeks having restricted entrances. Some reduction in invertebrate populations of the tidal creeks may have resulted from the herbicide application, though this impact may have been secondary.

Elser, H.J. 1966. Status of aquatic weed problems in Tidewater Maryland, spring, 1965. West Va. Pulp Paper Chem. Div. Taste Odor Control J. 32(8):1-6.

Aquatic weed control can be affected by chemical, physical, biological and mechanical methods of control. Nuisance characteristics and growth status of the following plants are discussed: *Myriophyllum spicatum*, *Trapa natans*, *Ulva lactuca*, *Ceratophyllum*, *Potamogeton perfoliatus*, *Elodea canadensis*, *Vallisneria americana* and *Zostera marina*.

Jaworski, N.A., D.W. Lear, Jr., and O. Villa, Jr. 1972. Nutrient management in the Potomac estuary, pp. 246-269. *In* G.E. Likens (ed.), Nutrients and eutrophication: the limiting nutrient controversy. Am. Soc. Limnol. Oceanogr. Inc., Lawrence, KA.

Because of the discharge of untreated or partially treated wastewater in the upper Potomac estuary of Maryland, water quality has become degraded. Studies beginning in 1965 have led to the formulation of a nutrient management program in this area.

Kolessar, M.A. 1967. Aquatic plants in Maryland--a growing menace. Proc. Amer. Soc. Civil Eng. 93(ww3):1-7.

Because of the increased population surrounding the Chesapeake Bay, the amount of pollutants entering the bay has increased. Some of the pollutants, such as herbicide run-off, add excessive nutrients. Plant populations, especially sea lettuce, water chestnuts and Eurasian water milfoil, have increased to a menacing level. Mechanical, chemical and biological control methods are discussed.

Lippson, A.J. ed. 1973. The Chesapeake Bay in Maryland: an atlas of natural resources. Johns Hopkins Univ. Press, Baltimore. 55pp.

A detailed atlas of the ecological factors affecting the Chesapeake Bay and the marine organisms living in bay waters.

Manning, J.H. 1965. The Maryland soft shall clam industry and its effects on tidewater resources. Univ. Maryland CBL Ref. No. 11. 25pp.

A thorough report on the soft shell clam industry including dredging techniques, the history and use of the clam in Maryland and the effects of the hydraulic clam dredge on tidewater resources.

Marsh, G.A. 1970. A seasonal study of Zostera epibiota in the York River, Virginia. Ph.D. dissertation, College of William and Maryland, 155pp.

The invertebrate macrofauna and epiphytes occurring on Zostera in the lower York River, Virginia, were sampled with the aid of SCUBA for 14 consecutive months. A collecting station was located at each of three different depths within a single eelgrass bed. Growth patterns of Zostera are discussed.

Marsh, G.A. 1973. The Zostera epifaunal community in the York River, Virginia. Chesapeake Sci. 14(2):87-97.

A quantitative description of the species composition, community structure and seasonal changes in the Zostera epifauna in a single eelgrass bed in the lower YOrk River, Virginia.

Marsh, G.A. 1976. Ecology of the gastropod epifauna of eelgrass in a Virginia estuary. Chesapeake Sci. 17:182-187.

Eelgrass provides a substrate for a highly diverse epibiotic community. Gastropods are the predominant faunal element. In the York River, Virginia, 10 species of prosobranch gastropods and 13 species of opisthobranch gastropods were found. The structure and species composition were studied.

Orth, R.J. 1971. Benthic infauna of eelgrass, Zostera marina, beds. M.S. Thesis, Univ. Virginia. 79pp.

The Zostera beds in the Chesapeake Bay, York River estuary and Chincoteague Bay, were sampled in March and July to determine the abundance and diversity of the infauna. A total of 117 macroinvertebrate taxa were found and most beds, except at the upper estuary limits, were similar.

Orth, R.J. 1973. Benthic infauna of eelgrass, *Zostera marina*, beds. Chesapeake Sci. 14(4):258-269.

In 1970, 117 macroinvertebrate taxa were collected from core samples of *Zostera marina* from the Chesapeake Bay York River estuary and from Chincoteague Bay. Seasonal differences were noted. The samples taken were compared for faunal dominance, similarity, diversity and composition. The environmental conditions, with emphasis on salinity and sediments were sampled and compared.

Orth, R.J. 1975. The role of disturbance in an eelgrass, *Zostera marina*, community. Ph.D. thesis, Univ. Maryland, College Park.

Eelgrass beds in the Chesapeake Bay were studied to determine: (1) how eelgrass affects community structure of associated infauna, and (2) responses of infauna communities to different levels of natural and artificially induced disturbances.

Orth, R.J. 1975. Destruction of eelgrass, *Zostera marina*, by the cownose ray, *Rhinoptera bonasus*, in the Chesapeake Bay. Chesapeake Sci. 16(3):205-208.

The cownose ray, *Rhinoptera bonasus*, digs deeply into water bottoms to feed on the hard shelled molluscs. The destruction of *Zostera marina* beds is attributed to this digging which uproots this ecologically important marine plant.

Orth, R.J. 1976. The demise and recovery of eelgrass, *Zostera marina*, in the Chesapeake Bay, Virginia. Aquatic Botany 2:141-159.

From 1971-1974 eelgrass, *Zostera marina*, L. declined 36%. Evidence indicating the loss was drawn from aerial photographs and ground truth reconnaissance. The deline is attributed to the cownose ray, human disturbance and a rise in water temperature.

Orth, R.J. 1977. The importance of sediment stability in seagrass communities, pp. 281-300 *In* B.C. Coull (ed.). Ecology of marine benthos. S. Carolina Press, Columbia.

Dense seagrass beds, such as *Zostera* in the Chesapeake Bay, stabilize sediments, promote diverse and abundant benthic fauna and protect fauna from predation from blue crabs.

Orth, R.J. In press. Effect of nutrient enrichment on the growth of eelgrass, *Zostera marina*, in the Chesapeake Bay, Virginia. Mar. Biol.

By adding two commercial fertilizers, *Zostera marina* was shown to be nutrient limited, competitively exclusive of *Ruppia maritima* and growth related to sediment nutrient supply.

Orth, R.J. and H. Gordon. 1975. Remote sensing of submerged aquatic vegetation in the lower Chesapeake Bay, Virginia. Final Report to National Aeronautics and Space Administration. Contract NASI-10720. 62pp.

Experimental Kodak water penetration film and black and white near infrared film were used in studying the submerged aquatic vegetation of the lower Chesapeake Bay. Between 1971 and 1974, there was a 36% reduction in the amount of vegetation.

Philipp, O.C. and R.G. Brown. 1965. Ecological studies of transition zone vascular plants in South River, Maryland. Chesapeake Sci. 6(2):73-81.

Two transition zone areas in the South River, Maryland, were studied to determine the distrubution of aquatic vascular plants. One area was near the mouth of the river and the other area was at the headwaters.

Rawls, C.K. and G.F. Beaven. 1963. Results of a 1962 field experiment subjecting certain estuarine animals to a 2,4-D ester. Proc. Southern Weed Conf. 16:343-344 (Abstr.).

Field studies in the Wicomico River (Potomac River estuary) of the effects of 2,4-D on *Mya arenaria*, *Crassostrea virginica*, *Callinectes sapidus* and *Lepornis gibbosus* resulted in no increase in normal mortality levels. Tissue assays revealed no 2,4-D residues in specimens.

Rawls, C.K. 1965. Field tests of herbicide tolicity to certain estuarine animals. Chesapeake Sci. 6(3):150-161.

Because of infestations of Eurasian water milfoil in the Chesapeake Bay and its tributaries since 1959, herbicide control with 2,4-D was necessary. Caged blue crabs (*Callinectes sapidus*), eastern oysters (*Crassostrea virginica*), soft shell clams (*Mya arenaria*) and various species were field tested with 2,4-D formulations to determine toxicity levels.

Rawls, C.K. 1975. Mechanical control of Eurasian water milfoil in Maryland with and without 2,4-D application. Chesapeake Sci. 16(4):266-281.

Between the late 1950s and 1964, Eurasian water milfoil increased from a few thousand acres to 200,000 acres. Milfoil grows in a variety of environments and its uses are minimal compared to its negative aspects. The herbicide 2,4-D was found to be effective in controlling milfoil; however, the required dosages could adversely affect the surrounding aquatic life. Because of this danger, smaller applications of 2,4-D plus mowing was recommended.

Rawls, C.K. In press. Food habits of waterfowl in the upper Chesapeake Bay, Maryland. 138pp.

Waterfowl gizzards representing 18 species of geese and ducks from the Chestertown area, Remington Farms and Colton Point were analyzed to determine the value of Eurasian water milfoil (*Myriophyllum spicatum*) as a food source. Out of 2,747 gizzards examined, over 78% of all food eaten was plant material.

Rawls, C.K. and P. McKee. 1964. Maryland's 1963 program for regulation and evaluation of 2,4-D applications. Proc. Southern Weed Conf. 17:306-307.

Due to increased Eurasian watermilfoil infestations in the Chesapeake Bay and its estuaries, the Water Pollution Control Commission approved the use of 2,4-D treatments to individual applicants.

Shima, L.J., R.R. Anderson, and V.P. Carter. 1976. The use of aerial color infrared photography in mapping the vegetation of a freshwater marsh. Chesapeake Sci. 17(2):74-85.

Aerial color infrared photographs taken of a freshwater marsh on the Patuxent River in the spring and fall were correlated with field surveys taken at the same time. Color fluctuations indicated different species, growth and vigor of plants and environmental conditions.

Southwick, C.H. 1972. Tentative outline for inventory of aquatic vegetation: *Myriophyllum spicatum* (Eurasian watermilfoil). Chesapeake Sci. 13(suppl): S174-S176.

A brief outline for the future inventory of *Myriophyllum spicatum* emphasizing the description of the species and its ecological habitat.

Southwick, C.H. and F.W. Pine. 1975. Abundance of submerged vascular vegetation in the Rhode River from 1966 to 1973. Chesapeake Sci. 16(1):147-151.

Surveys on the distribution of redhead grass (*Potamogeton perfoliatus*), Eurasian watermilfoil (*Myriophyllum spicatum*), widgeongrass (*Ruppia maritima*), horned pondweed (*Zannichellia palustris*), sago pondweed (*Potamogeton pectinatus*) and elodea (*Elodea canadensis*), showing irregular declines, disappearances and changes in species dominance from 1966 to 1973.

Steenis, J.H., E.W. Ball, V.D. Stotts, and C.K. Rawls. 1967. Pest plant control with herbicides, pp. 140-148 *In* Proc. of the Marsh and Estuary Mgmt. Symp. Louisiana State Univ., Baton Rouge.

Herbicides are used in the marine environment to modify plant composition and density for improvement and preservation of the estuarine habitat. Because of the possible effects of the addition of herbicides, they should be screened for the development of control procedures, their characteristics should be more clearly defined and application rated should be checked.

Steenis, J.H. and V.D. Stotts. 1961. Progress report on control of Eurasian watermilfoil in Chesapeake Bay. Proc. Northeastern Weed Control Conf. 15:566-570.

silvex and 2,4-D herbicides have been used for control. Studies were conducted to determine how to apply these herbicides in tidal situations without detrimental effects to other marine life.

Steenis, J.H. and V.D. Stotts. 1965. Tidal dispersal of herbicides for controlling Eurasian watermilfoil in the Chesapeake Bay. Proc. Southern Weed Conf. 18:507-511.

Because Eurasian watermilfoil has become a national weed problem, more effective tidal dispersal control with herbicides is necessary. 2,4-D, currently widely used, was considered to be effective. Paraquat and particularly diquat are excellent herbicides for milfoil control under tidal conditions but are less well understood than 2,4-D.

Steenis, J.H., V.D. Stotts, and C.R. Gillette. 1962. Observations on distribution and control of Eurasian watermilfoil in Chesapeake Bay, 1961. Proc. Northeastern Weed Control Conf. 16:442-448.

Eurasian watermilfoil is an adaptable aquatic plant that grows in fresh and saline waters. Boat surveys were made to determine the extent of infestation. Control of the plant was tested using 2,4-D.

Steenis, J.H., V.D. Stotts, D.S. Haven, and A.A. Whipp. 1964. Developments on control of Eurasian watermilfoil in the Chesapeake Bay region, 1963. Proc. Southern Weed Conf. 17:321-323.

Although 2,4-D can be used effectively to control the spread of Eurasian watermilfoil, the period of application is limited to the last ten days of May and the first week of June in the Chesapeake Bay. Lengthening the application period and using water movement for dispersal are discussed.

Stewart, R.E. 1962. Waterfowl populations in the upper Chesapeake region. U.S. Fish Wildl. Serv. Spec. Sci. Rept. Wildl. No. 65. 207pp.

Annual and seasonal variations in the numbers and kinds of waterfowl in 13 major habitat regions of the upper Chesapeake Bay are accounted for from 1953 to 1958. Data from U.S. Bureau of Sport Fisheries and Wildlife aerial waterfowl populations, observations of breeding, migration, and the gullet and gizzard analyses of 1,240 specimens are presented.

U.S. Army Corps of Engineers, Baltimore District. 1974. Chesapeake Bay, existing conditions report. Vol. 1-7.

Comprehensive investigation of Chesapeake Bay resources for baseline data on present status.

U.S. Army Corps of Engineers, Baltimore District. 1977. Chesapeake Bay, future conditions report. Vol. 2-12.

Investigation of present and future conditions of the Chesapeake Bay with recommendations for future social, economic and ecological criteria.

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